Environmental Risk Assessment for Peninsula Harbour Area of Concern

Final Report
Revision 2

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ACRONYMS AND ABBREVIATIONS

%	percent
2,3,7,8-TCDD	2,3,7,8-tetrachlorodibenzo-*p*-dioxin
AhR	arylhydrocarbon receptor
AOC	Area of Concern
AUF	area use factor
BCF	bioconcentration factor
BEAST	BEnthic Assessment of SedimenT
BMF	biomagnification factor
BSAF	biota sediment accumulation factor
BW	body weight
CCME	Canadian Council of Ministers of the Environment
CDF	confined disposal facility
Ci	dietary concentration
cm	centimetre
COA	Canada-Ontario Decision-Making Framework for Assessment of Great Lakes Contaminated Sediment
COC	chemical of concern
CSM	Conceptual Site Model
day/year	day per year
DFO	Department of Fisheries and Oceans
DI	daily intake
EC50	effect concentration for 50% of population tested
EDI	estimated daily intake
Environ	ENVIRON International Corporation
EPC	exposure point concentration
ERA	ecological risk assessment
fE	fraction of equilibrium attained at the time of consumption
FI	fraction ingested from source
FI:bw	food ingestion-to-body weight ratio
FIR	food ingestion rate
FMR	free metabolic rate
g	gram
g/day	gram per day
g/g-day	gram per gram per day
GIS	geographic information system
GLWQB	Great Lakes Water Quality Board
ha	hectare
HC	Health Canada
HHRA	human health risk assessment

HQ	hazard quotient
IJC	International Joint Commission
ISQG	interim sediment quality guideline
JC	Jellicoe Cove
kcal/kJ	kilocalorie per kilojoules
kg	kilogram
kg/day	kilogram per day
kJ/day	kilojoules per day
km	kilometre
LEL	lowest effect level
LOAEL	lowest observed adverse effect level
LOEL	lowest observed effect level
m	meter
m/s	meter per second
mg/kg	milligram per kilogram
mg/kg-day	milligram per kilogram body weight per day
MNR	Ministry of Natural Resources
MOE	Ministry of the Environment
MOEE	Ministry of the Environment and Energy
n	sample size
ng/g	nanogram per gram
ng/kg	nanograms per kilogram
NOAEL	no observed adverse effect level
NRC	National Research Council
Pa	Pascal
PAC	Public Advisory Committee
PAH	polycyclic aromatic hydrocarbon
PCB	polychlorinated biphenyl
PEL	probable effect level
PQRA	preliminary quantitative risk assessment
RAP	Remedial Action Plan
RC	reference concentration
RPH	rest of Peninsula Harbour (i.e., excluding Jellicoe Cove)
SDB	Standards Development Branch
SEL	severe effect level
SFCMP	Sport Fish Contaminant Monitoring Program
SWAC	spatially weighted average concentration
TDI	tolerable daily intake
TEF	2,3,7,8-TCDD toxic equivalence factor
TEQ	2,3,7,8-TCDD toxic equivalent
TOC	total organic carbon

TRG	tissue residue guideline
TRV	toxicity reference value
UCLM	upper confidence limit of the mean
UF	uncertainty factor
µg/goc	micrograms per gram organic carbon
USEPA	United States Environmental Protection Agency
VEC	valued ecosystem component
WHO	World Health Organization

EXECUTIVE SUMMARY

ENVIRON evaluated risks posed by mercury and polychlorinated biphenyls (PCBs) in the Peninsula Harbour Area of Concern (AOC). This work was conducted to aid Environment Canada, Ontario Ministry of the Environment, Marathon Pulp Inc., EcoSuperior, the public, and other stakeholders in understanding whether sediment management is warranted to protect human health and the environment. To facilitate management decisions, the AOC is divided into two areas: Jellicoe Cove (JC) and the rest of Peninsula Harbour (RPH). Because there is no physical barrier separating JC and RPH, water, sediment and/or biota are able to move freely between the two areas.

Project Objectives

1. Estimate ecological and human health **risks** posed by mercury and PCBs in sediment and biota in Peninsula Harbour.
2. Develop numerical **sediment management goals** based on risks, guidelines, background, and source control measures.
3. Estimate **area and volume of sediment** warranting management in order to achieve the sediment management goals.
4. Predict **residual risks** that would remain following source control measures.

This environmental risk assessment includes three main parts: 1) an ecological risk assessment (ERA) that focuses on the potential for adverse effects in benthic organisms, fish and wildlife; 2) a screening level human health risk assessment (HHRA) that focuses on the potential for adverse effects in people; and 3) risk management that defines cleanup goals and areas and volumes of sediment warranting management under different scenarios.

The first part—the ERA—evaluates risks to four types of ecological receptors: benthic invertebrates (sediment dwelling organisms), fish, piscivorous (fish-eating) birds, and piscivorous mammals. For the latter three groups, representative species are employed as surrogates for the full range of wildlife species that likely inhabit the area. Those surrogates were selected based on their exposure potential and toxicological sensitivity. Findings for these surrogates are expected to be protective of other species in the receptor groups.

- Based on multiple lines of evidence, risks posed by mercury and PCBs to benthic invertebrates are not significant.
- Comparisons of tissue concentrations to TRVs indicate that reproduction may be impaired in sportfish (e.g., lake trout, walleye, lake whitefish) and bottom dwelling fish (e.g., longnose sucker) in the AOC due to mercury and, in the case of longnose sucker, PCBs. The potential for adverse effects is greatest in longnose sucker, where reproductive impairment may propagate to population level impacts. Although the potential for population level impacts in sportfish is less clear, the most highly exposed individual lake trout, walleye, and lake whitefish are predicted to be adversely affected. With the exception of longnose sucker, most fish species are not predicted to be adversely affected by PCBs. Available biological data are insufficient to confirm or refute conclusions based on comparisons of tissue concentrations to TRVs.
- Risks posed to common loons and other waterfowl by mercury and PCBs are not significant.

- Current concentrations of mercury in fish may reduce reproductive success in individual bald eagles and other piscivorous raptors foraging primarily within JC (i.e., if 75 percent [%] to 100% of prey are derived from JC). Risk estimates are not so high as to suggest acute toxicity or population level effects. Given the substantially greater area of RPH, bald eagles and other piscivorous raptors are expected to derive the majority of their prey from RPH. Thus, any adverse effects on bald eagles or other piscivorous raptors are unlikely to have population level consequences. Current concentrations of PCBs in fish do not pose a significant risk to piscivorous raptors foraging within the AOC.
- Current concentrations of mercury in fish do not pose a significant risk to mink or other piscivorous mammals foraging with the AOC. Current concentrations of PCBs in fish pose a significant risk to mink and other piscivorous mammals foraging within either RPH or JC and RPH combined. Risk estimates are not so high as to suggest acute toxicity or population level effects. A survey of habitat suitability along the shore of Peninsula Harbour found that constraints related to vegetative cover likely limit the size of the local population of mink and other piscivorous mammals, regardless of concentrations of PCBs in fish. Thus, population level effects in piscivorous mammals are unlikely.

The second part – the screening level human health risk assessment (HHRA) – uses site-specific information on the fishing behaviours of local residents to predict whether people are likely to be adversely affected by mercury or PCBs in the fish they catch and eat from Peninsula Harbour.

- A survey of residents of Marathon and the Pic River Reserve was conducted in order to determine how much fish people eat from Peninsula Harbour. 241 surveys were returned, with most households (84%) reporting consumption of sport-caught fish and a significant number (17%) reporting consumption of fish caught in Peninsula Harbour. There is no evidence that the presence of a fish consumption advisory for Peninsula Harbour influences fishing and eating decisions. The survey also found no evidence of subsistence fishing in Peninsula Harbour. People who eat the most sport-caught fish tend to fish waterbodies other than Peninsula Harbour, suggesting that Peninsula Harbour is not a targeted destination for avid anglers. While households reported eating many different kinds of fish, only one person reported eating longnose suckers. Thus, longnose sucker tissue concentrations are not considered in the evaluation of risks to anglers from fish consumption. The fish consumption survey indicates that 80% of the respondents eat 17.5 g/day of sport-caught fish or less; that rate is used in the screening level HHRA to estimate dose. The fraction of the total sport-caught fish consumed that is from Peninsula Harbour was estimated using the median value (0.2) reported by anglers fishing at least some of the time in the AOC.
- Current concentrations of methylmercury in fish tissue do not pose a significant risk to adult sport anglers or household members. In contrast, current concentrations of PCBs in fish tissue pose a significant risk to adult sport anglers and household members (i.e., toddlers, children, and adolescents).

The third part – risk management – involves making decisions based on sediment assessment results. Sediment management goals are concentrations of mercury and PCBs in sediment that warrant consideration for such decisions. Although goals derived from acceptable risk benchmarks, guidelines, and background fish tissue levels were calculated, those related to acceptable risk benchmarks most directly reflect the conclusions of the risk assessment and are, therefore, most pertinent.

- The **risk-based sediment management goal for methylmercury is 0.0020 mg/kg** (protective of fish), while current spatially weighted average concentrations of methylmercury in JC and RPH surface sediment are 0.0051 mg/kg and 0.0019 mg/kg, respectively. Thus, the current concentration of methylmercury in RPH does not exceed the risk-based sediment management goal and the current concentration of methylmercury in JC is 2.6-fold higher than the goal.
- The **risk-based sediment management goals for PCBs are 0.06 mg/kg (protective of mink) and 0.19 mg/kg (protective of sport anglers).** Current spatially weighted average concentrations of PCBs in JC and RPH surface sediment are 0.14 mg/kg and 0.12 mg/kg respectively. Thus, the current concentration of PCBs in JC exceeds the mink sediment management goal by 2.3-fold and does not exceed the sport angler sediment management goal. The current concentration of PCBs in RPH exceeds the mink sediment management goals by 2-fold and does not exceed the angler sediment management goal.

Management of sediment to achieve the risk-based sediment management goals would require extensive remediation in both JC and RPH. Additionally**, the existing hot spot of mercury and PCBs in surface sediment of JC serves as an ongoing source of contamination to the rest of JC and RPH.** Therefore, hot spot management for purposes of source control warrants evaluation.

Source control options focus on total mercury, rather than methylmercury or PCBs. Total mercury acts as a source for methylmercury generation and rapid turnaround analysis and field screening measurements (which are necessary during remedial activities) are far more feasible for total mercury than for methylmercury. The PCB hotspot is generally co-located with the total mercury hotspot, but is somewhat smaller. Therefore, delineation based on total mercury captures much of the area with elevated concentrations of PCBs. The converse is not true: delineation based on PCBs misses much of the area with elevated concentrations of mercury.

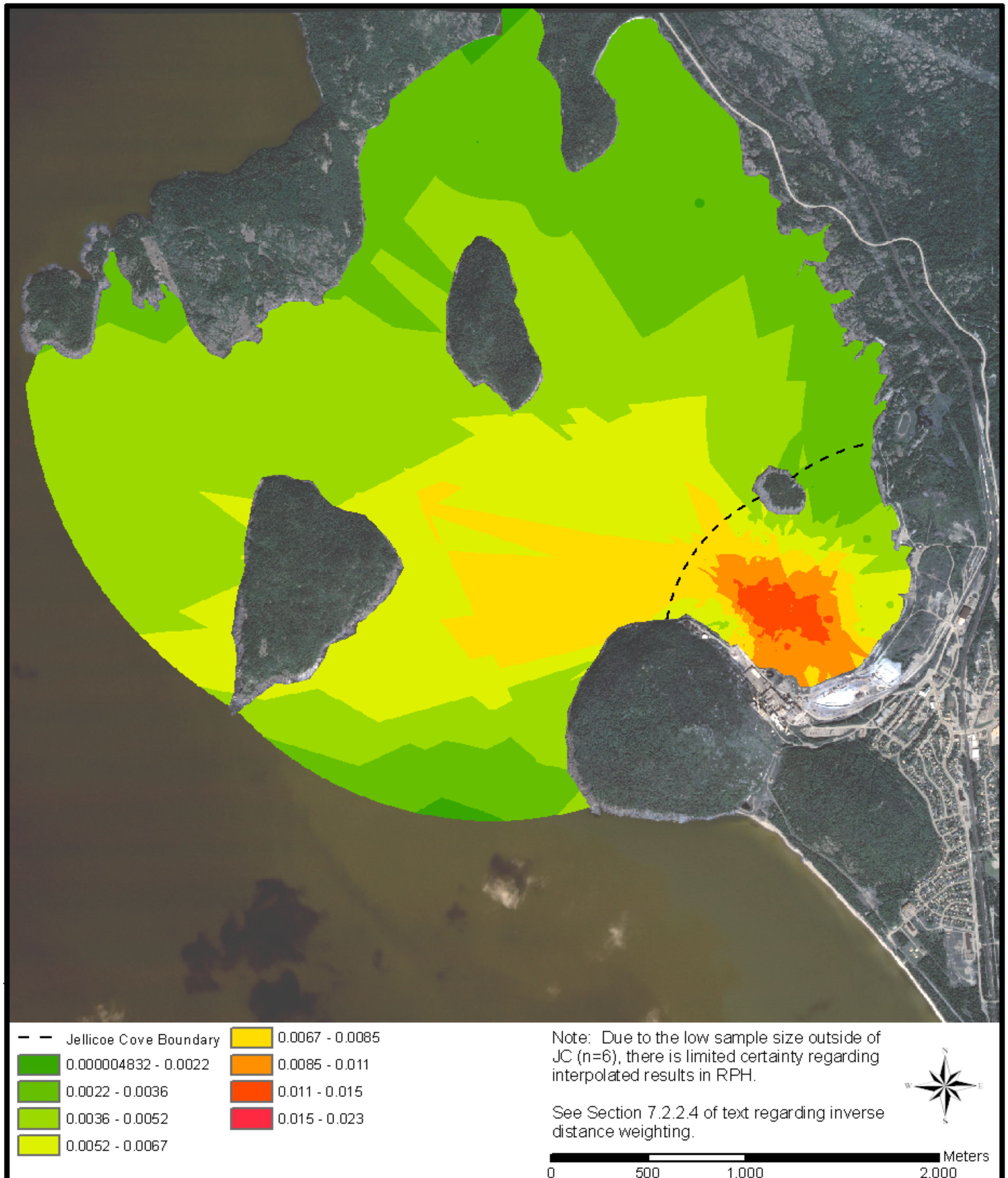
Table ES-1. Area and Volume of Sediment Warranting Management to Achieve Hot Spot-Based Sediment Management Goals

Hot Spot-Based Sediment Management Goal (Total Mercury, mg/kg)	Area of Jellicoe Cove with Concentrations Above Goal (m^2)	Average Depth of Sediment in Area[a] (cm)	Volume of Sediment Associated with Area (m^3)[b]	Residual Hazard Quotients	
				Longnose Sucker[c]	Mink[d]
2	250,000	17	42,500	1	1
3	182,000	18	32,760	1	1
6	83,000	18	14,940	1	2
10	39,000	20	7,800	1	2

a. Average sediment depth data based on Beak (2000).
b. Dredging costs estimated to range from $2.3 million to $13 million, based on rule-of-thumb estimate of $300/cubic meter (NRC 2001).
c. Assumes longnose sucker inhabit Jellicoe Cove 25% of the time and the rest of Peninsula Harbour 75% of the time.
d. Assumes mink forage within Jellicoe Cove 25% of the time and the rest of Peninsula Harbour 75% of the time.

Table ES-1 lists the areas and volumes of sediment targeted for source control through management of hot spots, based on varying definitions of the hot spot boundary. Figure ES-1 maps the distribution of methylmercury in the AOC, while Figure ES-2 maps the overlap between the total mercury hot spots and methylmercury concentrations in JC.

Although hot spot remediation scenarios are based on source control, rather than risk mitigation, management of sediment containing elevated concentrations of mercury and total PCBs is expected to reduce risks. In particular, hot spot remediation is predicted to reduce HQs for fish from methylmercury by 7% to 62%, depending on the mercury concentration used to define the hot spot and the fraction of time that fish are assumed to forage in JC and RPH. If the hot spot is defined by total mercury in sediment greater than either 2 mg/kg or 3 mg/kg, residual HQs for fish are not predicted to exceed the target HQ of 1, regardless of the amount of time that fish are assumed to forage in JC and RPH. Hot spot remediation is predicted to reduce HQs for mink from total PCBs by 4% to 44%, depending on the mercury concentration used to define the hot spot and the fraction of time that mink are assumed to forage in JC and RPH. Mink HQs do not exceed 1 when the hot spot is defined by 2 mg/kg or 3 mg/kg, regardless of the proportions of prey derived from JC and RPH. In summary, if the hot spot is defined by the total mercury concentration of 3 mg/kg in sediment, following remediation, methylmercury concentrations in JC are expected to be half of current levels (i.e., reduced from 0.0052 mg/kg to 0.0027 mg/kg) and risks are expected to be reduced to acceptable levels for fish and mink.



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Spatial Distribution of Methylmercury Concentration in AOC Surface Sediments

Figure ES-1

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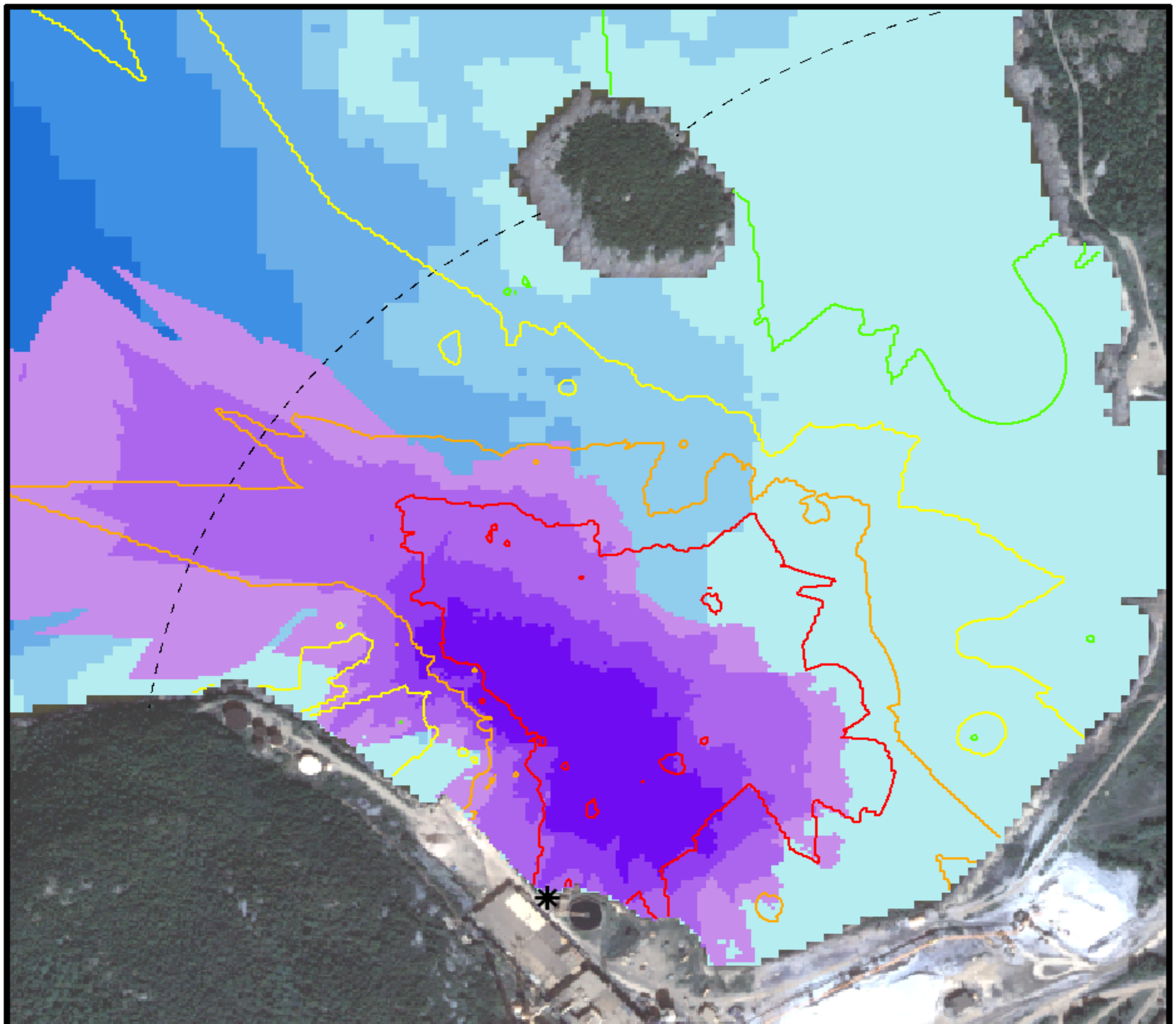

✱ Chlor-Alkali Outfall

— — Jellicoe Cove Boundary

Methylmercury Concentration (mg/kg)
- 0.004 mg/kg
- 0.006 mg/kg
- 0.008 mg/kg
- 0.01 mg/kg

Total Mercury Concentration
- Concentration > 2 mg/kg
- Concentration > 3 mg/kg
- Concentration > 6 mg/kg
- Concentration > 10 mg/kg

Depth (m)
- 1.9 - 9.3
- 9.4 - 17
- 18 - 23
- 24 - 31
- 32 - 39
- 40 - 47
- 48 - 57
- 58 - 62

Meters
0 62.5 125 250 375 500

See Section 7.2.2.4 of text regarding inverse distance weighting.

Ontario Ministry of the Environment's preliminary sediment quality guideline severe effect level for total mercury is 2 mg/kg.

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Total Mercury and Methylmercury
Hot Spots in Jellicoe Cove

Figure
ES-2

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1 INTRODUCTION

ENVIRON International Corporation (ENVIRON) evaluated risks associated with mercury and polychlorinated biphenyls (PCBs) in the Peninsula Harbour Area of Concern (AOC) on behalf of Environment Canada, pursuant to Contract #KW405-06-0213 (executed October 11, 2006) and amendments. This environmental risk assessment supports the development of a sediment management strategy for Peninsula Harbour. In particular, this work was conducted to aid Environment Canada, Ontario Ministry of the Environment (MOE), Marathon Pulp Inc., EcoSuperior, the public, and other stakeholders in understanding whether sediment management is warranted to protect human health and the environment.

The overall project objectives are to: 1) estimate risk posed by mercury and PCBs in Peninsula Harbour sediment and biota to both ecological and human receptors; 2) develop numerical sediment management goals based on the findings of the risk assessment, existing guidelines, background concentrations, and source control of hot spots; 3) estimate the area and volume of sediment requiring management in order to achieve these sediment management goals; and 4) predict residual risks that would remain following several different management scenarios.

This report is organized as follows. This Section (Section 1.0) presents overviews of risk assessment, the AOC, and studies that have been conducted on the AOC. Section 2.0 describes the species that are the focus of the ecological risk assessment (ERA) and the endpoints evaluated in the ERA. Section 3.0 describes how exposure of the ecological receptors is characterized, while Section 4.0 describes the ecotoxicity of the AOC's two main contaminants, mercury and polychlorinated biphenyls (PCBs). Section 5.0 integrates information on exposure and ecotoxicity to yield findings related to ecological risks posed by mercury and PCBs. Section 6.0 evaluates the potential for adverse effects in people that eat fish caught in Peninsula Harbour. Section 7.0 discusses sediment management options based on the findings of the risk assessment. Section 8.0 summarizes the overall report. References are listed in Section 9.0. This risk assessment includes a number of appendices, as well. Appendix A is a report on sediment sampling conducted in 2007. Appendix B is the database of chemistry results used in the risk assessment. Appendix C details practices employed in managing the analytical data. Appendix D describes the methods and results of a mink and otter habitat survey conducted in 2007. Appendix E details the food web model used to relate concentrations of mercury and PCBs in sediment and in fish tissue. Appendix F describes the methods and results of a local survey of fish consumption conducted in 2007.

1.1 Overview of Risk Assessment

MOE (2005a) states that risk "is a measure of the probability that a hazard will cause harm to an individual, population or the natural environment under defined conditions of exposure to a contaminant. Risk assessment is the scientific examination of the nature and magnitude of risk. It is a scientific process used to describe and estimate the likelihood of adverse health effects resulting from exposure of both human and ecological receptors to environmental contaminant(s)." Elements of an environmental risk assessment that focus on benthic invertebrates, fish and wildlife comprise the ERA, while those elements that focus on humans comprise the screening level HHRA. Elements of risk management, such as the development of sediment management goals and delineation of areas exceeding those goals, may also be included in an environmental risk assessment.

The Ministry of Environment and Energy (MOEE) (1996a) describes risk with the following expression:

Eqn. 1

$$\text{Risk} = \text{Severity of Event (Hazard)} \times \text{Exposure}$$

As discussed in Barnthouse et al. (2008), with respect to ecological receptors, "[r]egulations, policies, directives, and guidance documents frequently discuss the need for [ERAs] to consider risks to populations, not simply to individual organisms or organism-level attributes. The reason for this is that, from a management perspective, the population-level attributes such as abundance, persistence, age composition, and genetic diversity are usually more relevant than are the health or persistence of individual organisms" (Barnthouse et al. 2008). Thus, if an ERA predicts unacceptable risks in ecological populations and communities, management actions are typically evaluated and often taken to mitigate such risks. Most often, however, ERAs evaluate only individual-level effects, due to technical challenges and cost of conducting population-level assessments. If an ERA predicts unacceptable risks in individual organisms, management actions generally consider the proportion of individual organisms at risk, the spatial scale of the impact, whether the species at risk are protected, and whether acute effects are predicted. In contrast with ERAs, if unacceptable risks are predicted for individual human receptors in an HHRA, then management actions are typically evaluated and often taken to mitigate such risks because society places a high value on individual human lives.

Key guidance documents considered in the preparation of this risk assessment include:

- Guidance on Site Specific Risk Assessment for Use at Contaminated Sites in Ontario (MOEE 1996a);
- Procedures for the Use of Risk Assessment under Part XV.1 of the Environmental Protection Act (MOE 2005a);
- Framework for Ecological Risk Assessment (United States Environmental Protection Agency [USEPA] 1992a);
- Ecological Risk Assessment Guidance for Superfund: Process for Designing and Conducting Ecological Risk Assessment (USEPA 1997a);
- Guidelines for Ecological Risk Assessment (USEPA 1998);
- Federal Contaminated Site Risk Assessment in Canada, Part I: Guidance on Human Health Preliminary Quantitative Risk Assessment (PQRA) (HC 2004a);
- Federal Contaminated Site Risk Assessment in Canada, Part II: Health Canada Toxicological Reference Values (TRVs) (HC 2004b);
- Human Health Risk Assessment for Priority Substances (HC 1994);

- Risk Assessment Guidance for Superfund, Volume I: Human Health Evaluation Manual (Part A) (USEPA 1989a);
- Assessing Human Health Risks from Chemically Contaminated Fish and Shellfish: A Guidance Manual (USEPA 1989b);
- Human Health Evaluation Manual, Supplemental Guidance: "Standard Default Exposure Factors" (USEPA 1991);
- Exposure Factors Handbook (USEPA 1997b); and
- Guidance for Assessing Chemical Contaminant Data for Use in Fish Advisories. Vol. 2. Risk Assessment and Fish Consumption Limits (USEPA 2000).

1.2 Overview of Peninsula Harbour AOC

This environmental risk assessment focuses on the Peninsula Harbour AOC, located on the north shore of Lake Superior near the town of Marathon, Ontario (Figure 1-1). The harbour is about 3 kilometres (km) wide and 4 km long and is bound by Ypres Point to the north, a peninsula to the south, Hawkins Island to the west, and the main shoreline to the east. The harbour is sheltered from open waters of Lake Superior by two islands (Hawkins Island and Blondin Island) and two peninsulas to the north and south. The only waterways that enter into Peninsula Harbour are two small creeks, Shack Creek and an unnamed creek north of Shack Creek. Shack Creek passes through a closed wood waste storage site before flowing into the harbour, while the unnamed creek passes through a second closed wood waste site.

Ice cover, currents, shoreline and relief influence general environmental conditions in the harbour. Freeze-up in Peninsula Harbour generally occurs in early December, while ice break-up usually occurs in mid- to late-April (MNR 1984). Jellicoe Cove (JC) is reportedly ice-free during the winter (BEAK 2000). Currents 6 meters (m) below the water surface generally flow from the southwest to northeast in open water areas of the harbour, with currents averaging about 0.04 meters per second (m/s) (Environmental Hydraulics Group 1993). Within JC, bottom currents (0.5 m above the bottom) primarily move to the west-northwest (Skafel 2006). Skafel (2006) deployed a Nobska MAVS single point time of flight acoustic current meter over a 13-month period. The mean bottom current during the deployment was 0.014 m/s and the maximum was 0.19 m/s. Skafel (2006) also considered shear stress, concluding that for smooth sand bottom, no events during the deployment of the bottom current meter were vigorous enough to exceed the estimated critical shear of 0.26 Pascal (Pa), given no current. The shear stress was estimated based on sediment cores tested in a rotary flume ex situ (Krishnappan and Biberhofer 2003, as cited by Skafel 2006). However, for coarser sediment, there were 16 bursts when shear exceeded the critical value, indicating potential sediment instability under conditions with strong currents.

Much of the shoreline of Peninsula Harbour is characterized by rugged, hilly terrain with complex and steep slopes and cliffs extending into the water. Relief in excess of 150 m is common (BEAK 2000). There are no significant coastal wetlands along the shoreline of Peninsula Harbour that would support extensive growth of aquatic vegetation. Hamilton (1987) reported a complete lack of aquatic macrophytes in the areas of Peninsula Harbour that he surveyed for fish habitat. Sparse bottom vegetation is illustrated in Photo 1-1. It is worth noting that this photograph was taken in October and, therefore, may not be representative of conditions during the peak growing season.

Photo 1-1. Photograph of Jellicoe Cove Bottom (courtesy of J. Biberhofer)

The harbour contains several coves, including Jellicoe, Beatty, and Carden. Because the majority of analytical data available for Peninsula Harbour pertain to JC and in order to facilitate effective risk management decisions for the cove, the AOC is divided into JC and the rest of Peninsula Harbour (RPH). Although the theoretical boundaries of JC and RPH are illustrated in Figure 1-1, there is not a physical barrier between the two areas. Water, sediment, and/or biota move between the two areas as a function of currents, wind, prey availability, habitat constraints, and other factors. Thus, the division of the AOC into JC and RPH is a simplification of the system that is intended to aid in management decisions. Area use factors (AUFs) are employed in this risk assessment to compare outcomes when biota are assumed to spend varying proportions of time foraging in JC and RPH and anglers are assumed to consume varying proportions of fish from the entire AOC, or JC only, or RPH only.

The bathymetry of the AOC is mapped in Figure 1-2, based on data provided by Environment Canada. The distribution of fine sediment (i.e., silt and clay) is mapped in Figure 1-3, based on all available grain size data for surface sediment (0-5 centimetres or cm and 0-10 cm). The fraction of total organic carbon (TOC) in surface sediment (0-5 cm and 0-10 cm) of the AOC is mapped in Figure 1-4.

RPH is approximately 1,070 hectares (ha) in area and has 19.1 km of shoreline. The maximum depth of RPH is approximately 37 m (Environmental Hydraulics Group 1993).

JC is approximately 97 ha in area. The median depth of JC is 12.5 m (Milani et al. 2002), while the maximum depth is 28 m (Eakins and Fitchko 2000). At the western edge, the depth of JC reaches approximately 20 m. As noted above, Skafel (2006) observed that bottom currents in JC primarily flow to the west-northwest, with mean and maximum currents of 0.014 m/s and 0.19 m/s, respectively. The total length of shoreline along JC (including the island) is 3.3 km. A large portion of the southeast shore is beach, with a coarse sand and gravel substrate. The Marathon Pulp Inc. facilities are located along the southwest shoreline of the cove. Portions of the shore adjacent to the mill have been armoured with large boulder/rubble material and a shipping wharf occupies some of the western shore. Bedrock shoreline occurs along the west and east heads of the cove. A boat launch and docks are located at the northeast corner of the cove. Although land adjacent to the cove is sparsely vegetated, aquatic macrophytes are fairly common in JC, including pondweed (*Potamogeton spp*.), waterweed (*Elodea sp*.) and stonewort (*Chara sp.*) (Eakins and Fitchko 2000).

In summary, the area of RPH is about ten times greater than JC. Of the total shoreline in the harbour, about 15 percent (%) occurs along JC, while about 85% occurs along RPH. On a purely mathematical basis, it is reasonable to assume that receptors are exposed to mercury and PCBs in RPH to a greater extent than they are exposed to mercury and PCBs in JC. However, unique features related to access and habitat could influence the relative use of the two areas. Such features are evaluated with respect to mink habitat quality, but not for other receptors. Because evaluation of such features in JC and RPH is beyond the scope of this risk assessment, area use factors (AUFs) are employed in this report to account for receptors' variable use of JC and RPH. AUFs are discussed further in Sections 3.3.1 and 6.2.2.4.

1.2.1 Operational History

The history of industrial activities in JC began with the opening of a bleached kraft pulp mill in 1946 in Marathon. During early mill operations, effluent was pumped into Lake Superior via an open channel. From the time that the pulp mill opened until 1983, effluent discharged directly to JC via the bark pond, barker drum, and wet drum overflow outfalls. Occasional power failures resulted in direct discharge of pulp mill effluent into JC through the main sump overflow. In 1984, a submerged diffuser was installed 350 m offshore of Pebble Beach. In 1995, an aerated stabilization basin located 5 km southeast of the mill began providing secondary treatment of mill effluent.

In 1952, a chlor-alkali plant was built adjacent to the mill to provide chlorine and caustic soda for use in the chlorine bleaching process at the pulp mill. The chlor-alkali plant used mercury as a mobile electrode in the chlorine manufacturing process. In 1972, a primary clarifier was constructed for all effluent streams except the bleach process. In August 1977, the chlor-alkali plant ceased operations. Although the chlor-alkali plant had closed in 1977, during the period from 1977 until 1984, chlor-alkali plant effluent continued to be treated to remove trace mercury. Treatment stopped in 1984, following the sealing and appropriate disposal of all mercury-contaminated equipment.

Water quality in JC was also likely impacted by the practice of log booming and log storage in the cove. While log storage ceased in 1983, log booming ceased in 1987, when it was replaced by rail and truck transport.

1.2.2 Regulatory History

The following regulatory history of the AOC is excerpted from BEAK (2000). Since 1973, the Great Lakes Water Quality Board (GLWQB) of the International Joint Commission (IJC) has identified specific areas of the Great Lakes with serious water pollution problems. Such areas were defined as geographical locations in the boundary waters where one or more of the generic IJC water quality objectives or jurisdictional standards or criteria were not being met and where beneficial uses were or could be impaired. The IJC identified Marathon-Peninsula Harbour as one of 69 problem areas of the Great Lakes. This designation was based on PCB concentrations in lake trout exceeding the Health and Welfare Canada fish consumption guideline of 2 milligrams per kilogram (mg/kg) and mercury concentrations in lake whitefish and lake trout exceeding the guideline of 0.5 mg/kg.

In 1981, in order to provide an ecosystem perspective, the GLWQB established AOCs based on environmental quality for all environmental media (i.e., water, sediment, biota) and to evaluate these areas with uniform criteria. Based on consideration of a number of criteria, the GLWQB classified AOCs into two categories: Class A—those areas exhibiting significant environmental degradation, where impairment of beneficial uses was deemed to be severe; and Class B—those areas exhibiting environmental degradation, where uses may be moderately impaired.

Peninsula Harbour was designated as a Class B AOC due to local aesthetic degradation resulting from foam and suspended solids near the mill shoreline outfall discharging to Lake Superior, as well as residual mercury contamination of sediments and biota possibly contributing to elevated concentrations in sport and commercial fish.

The IJC reported that mercury concentrations in sediment were substantially greater than the provincial guideline for open water disposal of dredged spoil in portions of the harbour due to historical discharges from the chlor-alkali plant. PCB concentrations exceeding the dredge spoil guideline were also found in some harbour sediments, possibly due to a previous point source discharge. Large lake trout (greater than 56 cm) and lake whitefish (greater than 30.5 cm) had mercury concentrations greater than the federal guideline for commercial sale and the provincial guideline for sport fish. PCB concentrations in locally netted fish (lake trout and suckers) also exceeded the federal guideline.

In 1985, in order to provide a more complete and coordinated assessment of AOCs, the GLWQB developed a more comprehensive assessment procedure. Using new criteria, the Great Lakes jurisdictions identified 42 AOCs, including Peninsula Harbour. In addition to the adverse aesthetic problems near the outfall due to foam and suspended solids, the IJC indicated that phenols in harbour waters, and heavy metals, including mercury, and low concentrations of PCBs occasionally found in sediments may adversely affect recreational fishing.

The A-B designation system lacked direction on how to track and measure progress in AOCs and how to remove a site from the AOC list. To address these limitations, the GLWQB adopted a new system of categories that represented a logical sequence for problem solving and resolution. The categories identified the status of the information base, programs that were underway to fill in information gaps, and the status of remedial efforts. Under the new system, Remedial Action Plans (RAPs) are developed by the responsible jurisdictions and submitted to the IJC for review. RAPs describe programs and measures which, when implemented, should mitigate the identified problems. A site is removed from the AOC list when evidence could be presented that the full complement of beneficial uses had been restored. If it is deemed not feasible to restore all uses, then the RAP identifies the quality and uses that can be achieved.

The Peninsula Harbour RAP Team – including representatives from Ontario Ministry of the Environment (MOE), Environment Canada, Ontario Ministry of Natural Resources (MNR) and Department of Fisheries and Oceans (DFO) – was formed in 1985 to develop a RAP for Peninsula Harbour. The process included the formation of a Public Advisory Committee (PAC) comprised of representatives from the Town of Marathon, Buchanan Forest Products, James River-Marathon Ltd. (now Marathon Pulp Inc.), Ontario Federation of Anglers and Hunters, Marathon Rod and Gun Club, Friends of Pukaskwa, and the Marathon & District Chamber of Commerce. One of the purposes of the PAC was to provide a basis for broad community support for RAP implementation.

Stage 1 and 2 RAP reports describe the environmental problems and beneficial use impairments for the AOC. Currently, the five beneficial use impairments in the Peninsula Harbour AOC are:

- Restriction on fish consumption;
- Degradation of fish populations;
- Degradation of benthos;
- Restrictions on dredging activities; and
- Loss of fish habitat.

1.2.3 Investigation History

Studies related to environmental conditions in Peninsula Harbour have been conducted by various government agencies and by private consultants since the late-1960s. Much of the following summary of historical investigations of the AOC is drawn from Golder (2006), as well as from the original studies. Additional detail on the various studies can be found in the original studies.

1.2.3.1 Peninsula Harbour Area of Concern, Environmental Conditions and Problem Definition RAP Stage 1 Report (1991)

The RAP Stage 1 Report consolidated the available information on Peninsula Harbour from the late 1960s to 1990 into a single source. It primarily identified existing concentrations of contamination in water and sediments and included some data on invertebrate community structure and fish tissue residues of mercury. No relationships were drawn between sediment mercury concentrations and fish tissue residues beyond basic inferences suggesting that sediment mercury was a source of mercury residues in fish tissue. Many of the studies related to fish tissue residues took place during plant operations or shortly after the closing of the plant. Thus, historical fish tissue residues likely reflect active discharges at the time, rather than releases from sediment.

1.2.3.2 Peninsula Harbour Sediment Study (Smith 1992)

MOE sampled sediment in Peninsula Harbour in 1991. Samples were analyzed for total mercury and methylmercury. Samples included surficial sediment grabs and six core samples (i.e., three from shallow areas and three from deep areas). Both total mercury and methylmercury concentrations in sediment within JC and adjacent areas of Lake Superior were elevated relative to background.

Based on a limited number of core samples, the study concluded that concentrations of mercury increased with depth in the JC hot spot area and suggested that there had been a net accumulation of sediments during the time period that the chlor-alkali plant was in operation.

1.2.3.3 Peninsula Harbour Flow Pattern Study (Environmental Hydraulics Group 1993)

To evaluate the feasibility of a capping remedy for JC, a flow pattern study was undertaken to assess water movement and determine which particle sizes would resist erosion. The study consisted of review of existing sediment data, as well as collection of additional physical data, such as grain size distribution and bathymetry, together with wind and current data (obtained primarily through a two-day drogue study).

The report concluded that local conditions in the harbour are sufficiently dynamic during certain periods that fine to very fine sands may be resuspended by wave action and transported to other areas. The report noted that the prevailing current during the time of the study (i.e., November) was southward, which would suggest that resuspended materials could be carried westward out of the embayment and into the open waters of Lake Superior.

1.2.3.4 Peninsula Harbour Remedial Action Plan. Stage 2: Remedial Strategies for Ecosystem Restoration (Peninsula Harbour RAP Team 2002)

The Stage 2 Report provided a brief review of existing conditions. A number of remedial options were identified to address the shallow water areas, where mercury concentrations in sediments exceeded 6.0 mg/kg. The report noted that, until new technologies become available that can provide cost-effective, long-term solutions to remediating the mercury-contaminated sediments in the entire AOC, the preferred course of action involved dredging and disposal of sediments from the area of highest contamination and allowing for natural recovery of the remaining area.

1.2.3.5 Peninsula Harbour Feasibility Study Phases 1 – 4 (BEAK 2000, 2001, 2002)

In 1999, BEAK commenced a feasibility study in Peninsula Harbour to determine the potential for combining the removal of contaminated sediments in JC with a marina construction project. The project involved the construction of a shore-based confined disposal facility (CDF) for the contaminated sediments, which would be capped with clean fill and used as the base for construction of the marina. As part of the study, a number of core samples were collected in JC using a grid pattern to determine the area and depth distribution of mercury. Those areas exceeding 6 mg/kg mercury were to be considered for removal and disposal in the CDF.

In Phase III of the study, BEAK (2002) conducted an assessment of the potential accumulation of mercury by fish in Peninsula Harbour under two scenarios: (1) no action; and (2) sediment removal. Based on existing sediment data, BEAK (2002) estimated that the current mean concentration of mercury in the sediments of Peninsula Harbour was 1 mg/kg, and that this could result in an average mercury concentration of 0.6 mg/kg in fish. The predicted concentration exceeded MOE guidelines for unrestricted consumption and the Health Canada (HC) regulatory limit of 0.5 mg/kg, but was similar to concentrations measured in fish from Peninsula Harbour.

1.2.3.6 Evaluation of Trends in Mercury Concentration in Sport Fish from Peninsula Harbour (Hayton 2002)

Hayton (2002) summarized the mercury tissue residue data in sport fish from Lake Superior collected as part of the MOE Sport Fish Contaminant Monitoring Program (SFCMP). Because the SFCMP has been collecting fish tissue residue data in Lake Superior since 1975, Hayton (2002) focused on identifying temporal trends in mercury concentrations in a variety of sport fish species. The three species upon which the assessment was focused were lake trout, lake whitefish, and longnose sucker.

Since 1975, fish tissue mercury concentrations have declined in all three species of fish collected in Peninsula Harbour. In 2002, only longnose sucker tissue residues exceeded any of the consumption restriction guidelines. Tissue residues of mercury in longnose suckers from Peninsula Harbour were approximately two-fold higher than those from other parts of Lake Superior. Of the two sampling stations located in Peninsula Harbour, one was located within JC. Lake trout were sampled from that JC station in 1997.

Because results are expressed on the basis of size-normalized fish, the data do not indicate if there has been a change in tissue residues in young fish proportional to the general decrease and also do not provide an indication of how tissue residues are distributed among size classes.

1.2.3.7 BEAST Assessment of Sediment Quality in Peninsula Harbour, Lake Superior (Milani et al. 2002)

In 2000, Environment Canada undertook additional sampling in Peninsula Harbour for analysis of sediment chemistry (including total and methylmercury), benthic community structure, and sediment toxicity. Toxicity tests with amphipods (*Hyalella azteca*), midges (*Chironomus riparius*), mayflies (*Hexagenia limbata*), and aquatic worms (*Tubifex tubifex*) were conducted using standardized Environment Canada biological test methods.

Although there was mortality of *H. azteca* at a few of sites, it was not mercury-related. The sites where mortality was highest all had low concentrations of mercury. In two cases, sediment type (i.e., sand in one case, clay in another) could have been the cause of the observed mortalities. Therefore, there was no measurable toxicity to *H. azteca* in response to mercury concentrations; mortality was attributable to habitat/substrate types.

In addition, the study concluded that there were no changes in benthic community structure that could be related to sediment mercury concentrations. Major differences in benthic communities occurred in areas where mercury concentrations were similar to background concentrations. Differences were attributed to sediment type and water depth. Similarly, toxicity was not directly related to sediment mercury concentrations. The study concluded that there were no apparent toxic effects on the benthic fauna as a result of sediment mercury contamination.

1.2.3.8 A Study of the Bioavailability of Mercury and the Potential for Biomagnification from Sediment in Jellicoe Cove, Peninsula Harbour (Grapentine et al. 2005)

In 2002, Environment Canada conducted additional investigations in Peninsula Harbour that focused on determining the potential for mercury biomagnification from JC sediments. The study assessed total mercury and methylmercury concentrations in sediment and two species of resident benthic organisms. The study included a number of sites in JC, as well as limited sites in RPH. A number of reference areas in Lake Superior, both south and west of Peninsula Harbour, were also evaluated. A strong relationship was observed between total mercury in sediment and invertebrates, with a significant (though weaker) relationship observed for methylmercury. Mercury concentrations in invertebrates were higher in JC than in reference areas, although the difference was more pronounced for amphipods than for midges.

The potential for mercury biomagnification in higher trophic level species (e.g., fish, piscivorous birds) was assessed using site-specific benthic invertebrate total mercury and methylmercury tissue residues, together with literature-derived biomagnifications factors, to predict tissue residues of mercury in fish and fish-eating wildlife. Tissue residues in fish were based on the assumption that the exposed fish feed exclusively in the JC area, and that 100% of their body burden is obtained through ingestion of invertebrates from this area. Overall, this study concluded that mercury is bioavailable and is being accumulated by benthic organisms in sediments in both JC and RPH. A group of sites in the southeastern section of JC was noted as contributing most consistently and significantly to potential bioaccumulation-related risks.

1.2.3.9 Water and Sediment Quality Monitoring Survey – Harbours and Embayments, Lake Superior and the Spanish River (Richman 2004)

In 1999, MOE undertook a study of harbours and embayments in Lake Superior that included Thunder Bay Harbour, Nipigon Bay, Jackfish Bay, Peninsula Harbour, and the mouth of the Pic River. Media sampled included water and sediment, which were analyzed for metals (including total mercury), nutrients, polycyclic aromatic hydrocarbons (PAHs), PCBs and organochlorine pesticides, and dioxins and furans.

1.2.3.10 PCB Investigations at Peninsula Harbour, Lake Superior (Hayton 2005)

Additional studies in Peninsula Harbour conducted by MOE in September of 2003 involved: sediment sampling, bioaccumulation in caged mussels, and forage fish analyses. Biota samples were collected from both JC and RPH and were analyzed for PCBs and total mercury. Sediment samples were collected from JC only and were analyzed for PCBs and total mercury.

Sediment samples included surface grabs and three core samples. Concentrations at depth were found to be greater than previously recorded, but were confined to the southwestern section of JC.

Young-of-year fish were collected for tissue residue analyses at three locations in Peninsula Harbour. Fish in the southern section of Peninsula Harbour had higher mercury tissue residues than fish in Carden Cove and both areas had higher tissue residues in young fish than at the control site (Neys Provincial Park).

Caged mussel biomonitoring samples were analyzed for mercury, accumulated over a three-week exposure period in September of 2003. Mussels were placed in the nearshore area in the southern part of the harbour, as well as at the control site in Neys Provincial Park. Mean mercury tissue residues in the mussels in JC were in the same range as those from the control site. However, mussels were not placed in the deeper hot spot areas, where higher sediment mercury concentrations would be expected to occur.

1.2.3.11 A Report on the Chemical and Physical Characteristics of the Sediment in Jellicoe Cove, Peninsula Harbour (Biberhofer and Dunnet 2005)

In 2003, Environment Canada undertook additional sediment sampling in JC. A total of 11 core samples were collected for mercury analyses, including total mercury for all core sections and methylmercury in the 0 to 5 cm interval. Cores for total mercury analyses were divided as follows: 0 to 1 cm, 1 cm to 3 cm, 3 cm to 5 cm, 5 cm to 10 cm, 10 cm to 15 cm, and 15 cm to 20 cm.

Analyses of sediment samples yielded results similar to those of previous surveys, with mercury concentrations increasing with sediment depth. The study found that there was a poor relationship between total mercury and methylmercury in sediment, but the relationship improved substantially when one station (with a predominantly gravel substrate) was excluded. Indeed, when corrected for sand dilution, the relationship between methylmercury and total mercury concentrations was very strong (R^2 = 0.82).

1.2.3.12 Review: Mercury and PCBs in Fish and Sediment in Jellicoe Cove, Peninsula Harbour (Sommerfreund et al. 2005)

Sommerfreund et al. (2005) reviewed existing studies in JC. While the study focused on the recent studies by BEAK (2002), Grapentine et al. (2003), Milani et al. (2002), Hayton (2005) and Biberhofer and Dunnet (2005), the review included data from the MOE studies of the 1970s.

The study included an analysis of spatial and depth distribution of mercury in the harbour, noting that the highest mercury concentrations occurred at depth in the western section of JC. Outside of this area, concentrations of mercury in sediment decreased rapidly. The study also noted that Biberhofer and Dunnet (2005) observed limited depth of isotope activity, which may be indicative of low net accumulation rates. Alternatively, the sediment profile may have been disturbed by episodic events.

Sommerfreund et al.'s (2005) key conclusions include:

1. The patterns of mercury and PCB concentration in sediment core profiles in JC are consistent with releases from the mill. Total mercury concentrations in sediment exceed the severe effect level (SEL) by three orders of magnitude and are greatest at greater than 10 cm depth close to the mill.

2. Historical total mercury in sediment is first subject to burial by relatively cleaner sediment. Contaminated sediment may have dispersed laterally towards the northwest and southeast.

3. Methylmercury concentrations in sediment are elevated in JC and generally parallel total mercury concentrations.

4. Mercury concentrations in pore water track those of particulate mercury and reductive dissolution of manganese oxides. The distribution of methylmercury in porewater is consistent throughout the core.

5. Total PCB concentrations in sediments exceed the lowest effect level (LEL) and are highest north of the mill.

6. Mercury and PCB concentrations have similar spatial distribution, suggesting a common source. Both are associated with fine-grained sediment, followed by movement through burial and lateral sediment transport.

7. Both total mercury methylmercury are transferred from sediment to benthic invertebrates in JC.

8. Mercury concentrations in several species of fish in Peninsula Harbour remain higher than those of Lake Superior and are above the consumption guidelines. Concentrations of mercury in longnose sucker, lake whitefish, and lake trout have declined considerably since 1975.

9. PCB concentrations in longnose sucker from JC exceed consumption guidelines. Concentrations of PCBs in longnose sucker have declined by one half from 1978 to 1990 and have remained stable since then.

1.2.3.13 Jellicoe Cove Sediment Sampling, January 2007

On January 10 through 12, 2007, DST collected surface sediment samples from 17 previously uncharacterized locations within JC. A report of field activities related to this sampling program is provided as Appendix A to this risk assessment. Work was conducted under subcontract to ENVIRON for the purpose of refining management areas identified in this risk assessment. Samples were analyzed for total mercury, methylmercury, PCBs, TOC, and grain size. While the resultant data on total mercury, methylmercury, PCBs and TOC are incorporated into this report, the grain size data are expected to be utilized in future analyses, such as the evaluation of remedial alternatives. Figure 1-3 presents the percent fines (silt and clay) for JC, as well as the rest of the AOC. The grain size data demonstrate that gravel is present (5% to 20%) in the sediment immediately adjacent to, and within about 100 m of the north and west sides of the JC peninsula. Between the JC peninsula and Skin Island, sediment is predominantly silt and clay. To the east and west of the cove, sediment has a higher percent sand.

1.2.3.14 Fish Sampling, August and September 2007

Environment Canada and MOE collected fish samples in August and September of 2007 for the purposes of refining risk estimates and reducing uncertainty in both the food web model and the risk calculations. In August 2007, fish samples were collected from two locations in JC by Environment Canada. One location was near Skin Island and the other was to the south-southwest of the island. One day set and one night gill-net set was deployed at each location. Longnose sucker and lake whitefish of appropriate sizes were submitted to ALS Laboratory Group (Edmonton, Alberta) and composited into samples comprised of two to four fish that were within 5 cm of each other in size. In total, 9 longnose sucker and 9 lake whitefish composite samples from JC were analyzed, 12 for PCB congeners and all 18 for total mercury. In September 2007, MOE performed gill net fish sampling in RPH. Longnose sucker, lake trout, and lake whitefish were collected, samples were filleted, and archived by MOE. Composite samples were prepared using fillets from fish within a 5-cm size range. Composites included one to three fish. Three lake trout, four whitefish, and four longnose sucker composite samples were analyzed; all 12 were analyzed for PCB congeners and 6 were analyzed for total mercury. The fish collected during these sampling events ranged in length from 20 cm to 55 cm.

1.3 Chemistry Data Summary

With the exception of the 2007 fish data (which were reported to ENVIRON by the laboratory, ALS Laboratory Group), Environment Canada provided ENVIRON with the data used in this risk assessment. These data relate to chemical analyses of sediment, surface water, and biota samples collected between 1997 and present. As detailed in Table 1-1, chemistry data used in this risk assessment include:

- Richman (2004): surface water and sediment samples collected in 1999;
- BEAK (2000, 2001): sediment samples collected in 2000;
- Milani et al. (2002): sediment samples collected in 2000;

- Grapentine et al. (2003): sediment and invertebrate tissue collected in 2002;
- Hayton (2002): fish tissue samples collected from the area of concern and nearby areas of Lake Superior since 1997;
- Hayton et al. (2005): sediment samples collected in 2003;
- Biberhofer and Dunnet (2005): sediment samples collected in 2003;
- Biberhofer and Dunnet (no date): sediment samples collected in 2005;
- DST (2007): sediment samples collected in 2007; and
- MOE and EC (2007): fish tissue samples collected in 2007.

Total mercury, methylmercury, and PCB analytical results from these studies were imported into an Access database (Appendix B). To the extent feasible and as appropriate, additional parameters (e.g., organic carbon, lipids, and metals) were also included in the database. All sampling data collected from within the AOC since 2000 were included in the assessment. That is, sampling data from within the AOC were not censored. Figure 1-5 illustrates all chemistry sampling locations considered in this report. Figures 1-6 and 1-7 illustrate the spatial distribution of total mercury and methylmercury concentrations in surface sediment in Peninsula Harbour. Figures 1-8 and 1-9 illustrate the spatial distribution of total PCB and 2,3,7,8-tetrachlorodibenzo-p-dioxin (2,3,7,8-TCDD) toxic equivalent (TEQ) concentrations in surface sediment in the harbour.

In order to depict the spatial distribution of mercury and PCBs in the AOC, a geospatial interpretive tool called inverse distance weighting was used to interpolate sediment concentrations in areas lacking sampling data. This tool estimates concentrations in unsampled areas based on the concentrations measured elsewhere, weighting those samples closest to each interpolated point more heavily than those that are more distant. In applying this tool, there is a trade-off between the relative weight given to more distant points and the degree of "structure" in the output (more weight assigned to distant points results in a smoother representation). Thus, to smooth the concentration isopleths, we increased the weight assigned to distant points. However, this practice created some areas where high concentrations some distance away influenced sparsely sampled areas, as in the case of the southwestern shore of Jellicoe Cove. Additionally, because sediment data in RPH are relatively limited, the accuracy of these figures is limited outside of JC. As with any interpretive geospatial tool, there is uncertainty in the interpolated concentrations at unsampled locations.

Certain data handling practices increased the applicability of the data for risk assessment purposes:

- Because fish tissue samples were analyzed for total mercury rather than methylmercury, all of the mercury measured in fish tissue was conservatively assumed to be methylmercury. As discussed in Section 4.1, methylmercury is the form of mercury that is most bioaccumulative and most toxic. Consequently, this assumption ensures the health-protectiveness of the risk assessment.

- The majority of fish tissue samples were analyzed as fillets, as is appropriate for use in screening level HHRA. However, whole body results more accurately represent ecological exposures. Therefore, fillet concentrations were converted to whole body concentrations for use in the ERA, using the procedures documented in Appendix C. The whole body fish tissue samples collected in 2007 were also used as reported and found to agree with the extrapolated results from fillet samples. Given the much greater number of fillet samples as compared to whole body samples, combining the two types of samples was judged to yield more certain results than would be provided by either the fillet dataset or the whole body dataset alone.
- Most fish tissue samples were collected from either very small fish (i.e., young-of-year) or relatively large fish (i.e., adults). Just three longnose sucker samples were composed of fish between 15 cm and 25 cm length. Medium-sized fish tend to be the size preferred by the wildlife receptors evaluated in the ERA. Therefore, fish assumed to be consumed by wildlife were size-normalized prior to calculating a representative concentration. The size normalization procedures are documented in Appendix C. For each wildlife receptor, the fish were normalized to the midpoint of the preferred prey size range.
- Fillet samples collected from fish of 15 cm in length or greater were used to evaluate consumption by humans.
- A variety of methods are available for the analysis of PCBs in environmental samples. Samples included in the database were analyzed for PCBs as total PCBs, homologues, or congeners. One sediment sample was analyzed for PCB and congeners, 000P8542 (C70533). That sample was also analyzed for total PCBs. Compared to other samples analyzed for total PCBs, 000P8542 (C70533) contained a relatively low concentration. Therefore, the sample would not provide a conservative or representative basis for estimating risks based on TEQs. Given this limitation, this risk assessment does not quantify TEQ risks, but instead discusses TEQ risks qualitatively as part of the ERA and screening level HHRA uncertainty analyses. Total PCB concentrations used in this assessment were analyzed by either total PCB methods or homologue-based methods. Although 18 of the fish samples collected in 2007 were analyzed for all PCB congeners, this relatively low sample size (as compared with the number of fish samples analyzed for total PCBs) indicates that a TEQ-based risk assessment approach would be less certain that the total PCB approach employed in this report.
- Spatially weighted average concentrations (SWACs) of mercury and PCBs in surface sediment (0 cm to 5 cm and 0 cm to 10 cm) were calculated using Thiessen polygons (Davis 1986, as cited in DOE 2006), as detailed in Section 7.2.1.

The data used in this risk assessment are compiled in Appendix B. The three tables in Appendix B are the components of the project database for the Peninsula Harbour AOC.

2 ECOLOGICAL RECEPTOR CHARACTERIZATION

As described by MOEE (1996a), receptor characterization is “the process of identifying the ecological (non-human) receptors of concern (VECs[1]), the effects against which it is desirable to protect the VECs, and the means or pathways specific to each VEC by which it may come into contact with contaminants.” Towards these ends, this section presents the ecological Conceptual Site Model (CSM) for the AOC, followed by a discussion of the selected VECs evaluated in the ERA and their respective life histories and characteristics that influence their potential exposure. Assessment endpoints – explicit expressions of the environmental values that are to be protected – are then defined. Because it is generally difficult or impossible to directly measure assessment endpoints, each assessment endpoint is evaluated through one or more measurement endpoints. This section closes with the selection of measurement endpoints used in the ERA.

2.1 Conceptual Site Model

A CSM is a written description and visual representation of predicted relationships between VECs and the chemicals of concern (COCs)[2] to which they may be exposed. MOE (2005a) describes the CSM as a physical description of the potential contamination problem from an ecological risk perspective. It draws upon the aquatic food web for the AOC, which is depicted in a simplified manner in Figure 2-1. The ecological CSM for Peninsula Harbour AOC is presented schematically in Figure 2-2. As shown, primary sources of COCs were historical discharges from the bleached kraft pulp mill and the chlor-alkali plant. Work is currently underway by MOE to determine whether additional sources of PCBs exist. The primary receiving media were Peninsula Harbour surface water and sediment. COCs in surface water preferentially partitioned to sediment, particularly the fine-grained sediment and those with relatively high TOC. While contaminated sediment has been buried by somewhat cleaner sediment, the sediment likely continues to function as a source to the water column and as a source of exposure to ecological receptors. Contaminated sediment is transported away from the area where it was originally deposited, due to the effects of bottom currents and bathymetry. Micro-organisms in sediment transform inorganic mercury in the sediment to methylmercury, a form that is more bioaccumulative and more toxic than other forms of mercury. Following uptake by aquatic organisms (i.e., benthic invertebrates, fish), biota tissue functions as a secondary source of exposure to upper trophic level receptors such as birds and mammals that eat benthic invertebrates and fish.

2.2 Valued Ecological Components

Valued Ecological Components (VECs) are resources or environmental features that: 1) are important to human populations; 2) have economic and/or social significance; 3) have intrinsic ecological significance; and/or 4) serve as a baseline from which the impacts of development

[1]Valued Ecosystem Components
[2]Total mercury, methylmercury and PCBs are defined as the COCs evaluated in this environmental risk assessment.

can be evaluated, including changes in management or regulatory policies (MOEE 1996a). Based on the ecological CSM, the VECs pertinent to this ERA include the benthic invertebrate community, fish populations, piscivorous bird populations [as represented by common loons (*Gavia immer*) and bald eagles (*Haliaeetus leucocephalus*)], and piscivorous mammal populations [as represented by mink (*Mustela vison*)]. In the cases of piscivorous bird and mammal populations, representative species were selected for quantitative evaluation based on both exposure potential and toxicological sensitivity (collectively referred to as susceptibility). While mink are commonly acknowledged to be highly sensitive to the toxicological effects of PCBs and mercury (e.g., Moore et al. 1999, Hornshaw et al. 1983), interspecies differences in sensitivity among avian species is less clear. Differences in exposure potential among species are qualitatively considered as a function of the fraction of aquatic prey in diet (higher fraction yields higher exposure) and body weight (lower body weight yields higher dose).

Because the representative species selected – common loons, bald eagles, and mink – are expected to be among the most susceptible of the species likely to inhabit the AOC, extrapolation of conclusions regarding these receptors are protective of other, less susceptible species. As such, common loons and bald eagles serve as conservative surrogates for species such as gulls (*Larus* spp.), ducks (e.g., *Anas* spp.), belted kingfishers (*Ceryle alcyon*), osprey (*Pandion haliaetus*), and herons (Ardeidae family), while mink serve as conservative surrogates for species such as raccoons (*Procyon lotor)*, otters (*Lutra canadensis)*, martens (*Martes Americana)*, and fishers (*Martes pennanti)*. The following subsections further describe the life histories of the selected VECs, focusing on those aspects of their behaviour that influence exposure to mercury, methylmercury, and PCBs in sediment and prey.

2.2.1 Benthic Invertebrate Community

The benthic invertebrate community lives in constant and direct contact with sediment that may be impacted by mercury and PCBs. Benthic invertebrates have vital functions within the ecosystem, including serving as a prey base for higher trophic level organisms and cycling of organic carbon and nutrients. Milani et al. (2002) conducted an extensive investigation of benthic invertebrate community composition in JC and other locations within Peninsula Harbour. Additionally, historical information on benthic invertebrate community composition is available from a detailed, species-level study of preserved invertebrate samples collected between 1969 and 1989 (Sibley et al. 1991). Milani et al. (2002) found that midge larvae (Chironomidae), oligochaete worms (Tubificidae and Naididae), fingernail clams (Sphaeriidae), isopods (Asellidae), and snails (Valvatidae) were prevalent, with other taxa (amphipods and oligochaetes) comprising less than 10% of the benthic community throughout the area surveyed. These results are compared to regional reference sites in Section 4.

2.2.2 Fish Community

As described by BEAK (2000), Peninsula Harbour supports a fish community that includes at least 31 species, of which 22 are native to Lake Superior. The community is dominated by coldwater species. Lake trout are stocked and have persisted as the dominant piscivorous fish species in the harbour.

BEAK (2001) conducted a fisheries resource and habitat assessment of JC and Carden Cove in August 2000, using gillnetting, seining, and electrofishing. Sixteen species of varying age classes were captured in JC. The most abundant species encountered in JC were longnose dace (*Rhinichthys cataractae*), longnose sucker (*Catostomus catostomus*), round whitefish (*Prosopium cylindraceum*), coho salmon (*Oncorhynchus kisutch*), mottled sculpin (*Cottus bairdii*), and slimy sculpin (*Cottus cognatus*).

Information on mercury and PCB concentrations in fish from Peninsula Harbour is available primarily for lake whitefish (*Coregonus clupeaformis*), lake trout (*Salvelinus namaycush*), longnose sucker, walleye (*Sander vitreus*), and pink salmon (*Oncorhynchus gorbuscha*). General life history characteristics related to their foraging behaviour and depth are presented below for each of these five fish species.

2.2.2.1 Lake Whitefish

As described on the FishBase website (http://www.FishBase.org/Summary/SpeciesSummary.php?id=234), lake whitefish is a member of the Salmonidae family that grows to a maximum 100 cm in length and 19 kilograms (kg) in weight. The lake whitefish is demersal (i.e., lives on or near the bottom and feeds on benthic organisms), anadromous (i.e., ascends rivers to spawn), and inhabits water depths ranging from 18 m to 128 m. It is primarily a lake dweller and appears to be rather sedentary, at least in the Great Lakes (Morrow 1980). Movement in large lakes generally consists of four stages: movement from deep to shallow water in the spring; movement back to deep water in the summer as the shoal water warms; migration to shallow-water spawning areas in the fall and early winter; and post-spawning movement back to deeper water (Morrow 1980). Discrete populations of lake whitefish form in large lakes (Morrow 1980). Schools of lake whitefish are local in their habits; tagging studies have shown that most individuals remain within 8 km of the tagging site (Becker 1983). Adults feed mainly on aquatic insect larvae, molluscs, and amphipods (Hart 1931, Koelz 1929), as well as other fishes and fish eggs, including their own (Scott and Crossman 1973). The only lake whitefish samples collected from the AOC were collected from RPH (i.e., none were collected from JC), possibly reflecting differences in habitat across JC and RPH. Lake whitefish are commercially fished and are targeted by recreational anglers.

2.2.2.2 Lake Trout

As described on the FishBase website (http://www.FishBase.org/Summary/SpeciesSummary.php?id=248), lake trout is a freshwater member of the Salmonidae family that is benthopelagic (i.e., foraging near the bottom as well as in midwater or near the surface), non-migratory and inhabits water depths ranging from 18 m to 53 m. The maximum (but atypical) reported length and weight of lake trout are 150 cm and 33 kg respectively. Page and Burr's (1991) observation that lake trout are found in both shallow and deep waters of northern lakes is supported by the collection of lake trout samples from both JC and RPH. A solitary wanderer, the extent of the lake trout's movements are apparently limited by the size of the lake and individual (Morrow 1980). Most lake trout live within approximately 80 km of preferred spawning grounds, although some large individuals may travel up to 400 km (Becker 1983). Although lake trout generally feed on a variety of organisms, such as freshwater sponges, crustaceans, insects, fishes (with a preference for ciscoes), and small mammals, some populations feed on plankton throughout their lives (Morrow 1980). Lake trout are commercially fished and are targeted by recreational anglers.

2.2.2.3 Longnose Sucker

As described on the FishBase website (http://www.FishBase.org/Summary/SpeciesSummary.php?id=2962), longnose sucker is a member of the Castostomidae family that grows up to 64 cm in length and 3.3 kg in weight. A demersal species, the longnose sucker tolerates pH values in the range of 6.5 to 7.8 and inhabits depths to 180 m. It is found in clear, cold, deep water of lakes and tributary streams (Page and Burr 1991). The longnose sucker moves from lakes into inlet streams or from slow, deep pools into shallow, gravel-bottomed portions of streams to spawn (Morrow 1980). While young fish remain in weed bed areas, older fish show offshore movement during daylight but tend to return to the same areas (Carlander 1969). Longnose suckers feed on benthic invertebrates (Scott and Crossman 1973). Young are consumed by other fish and fish-eating birds, while adults in spawning streams may be consumed by mammals, eagles, and osprey (Scott and Crossman 1973). Although longnose suckers are of minor commercial importance, they are targeted by some recreational anglers.

2.2.2.4 Walleye

As described on the FishBase website (http://www.FishBase.org/Summary/SpeciesSummary.php@id=3516), walleye is a member of the Percidae family that can grow to more than 100 cm and 11 kg in size and can live up to 29 years, although 7 years is a more normal lifespan (Becker 1983). Walleye are demersal and potamodromous (i.e., migratory within rivers). They occur in lakes and in medium to large rivers and prefer shallow, turbid habitat, but may occur in habitats as deep as 27 m. Although the preferred habitat of walleye would thus appear to be consistent with conditions within JC, no walleye samples are available for JC. Rather, all walleye samples were collected from RPH. Walleye feed at night, mainly on insects and fish. They also consume other prey (e.g., crayfish, snails, amphibians, small mammals) when fish and insects are scarce. Walleye can travel considerable distances in spring and fall (e.g., 20 km to 50 km); their movements are less extensive in summer (Becker 1983). Walleye are commercially fished and are targeted by recreational anglers.

2.2.2.5 Pink Salmon

As described on the FishBase website (http://www.FishBase.org/Summary/SpeciesSummary.php@id=240), pink salmon is a member of the Salmonidae family that can grow up to 76 cm in length and 6.8 kg in weight. Pink salmon are demersal and anadromous. They typically live for two years, dying shortly after spawning. According to the Michigan Department of Natural Resources (http://www.michigan.gov/dnr/0,1607,7-153-10364_18958-45686--,00.html.), this Pacific species was accidentally introduced to the Great Lakes in the mid-1950s and is now established in Lakes Superior, Huron, and Michigan. In the Great Lakes, pink salmon feed on a variety of fish and other aquatic animals. They migrate to streams for spawning in the summer, with the young returning to the lake the following spring. Distances traveled in the Great Lakes are not known, but Becker (1983) notes "much wandering." Adults can inhabit quite deep water (up to 250 m). Pink salmon samples (likely young-of-year, based on their small size) were collected from both JC and RPH. Great Lakes pink salmon are rarely caught by anglers, although some are taken while ascending streams. In general, the species is considered of high commercial importance.

2.2.3 Piscivorous Bird Populations

As previously noted, the VEC of piscivorous bird populations is represented by two susceptible species inhabiting the AOC, common loons and bald eagles. Because the two species differ in their preferred prey and foraging strategies, their exposure potentials are also expected to differ.

2.2.3.1 Common Loon

The common loon is a long-bodied, low-slung diving bird in the taxonomic order Gaviiformes. Common loons are large, ranging in weight from 2.5 kg to 6.1 kg, with males larger and heavier than females. In a three year study conducted by Barr (1986), male common loons from northwestern Ontario averaged 4.4 kg [sample size (n) = 23], while females averaged 3.54 kg (n = 15).

While most loons winter along coastal marine waters, they breed along inland freshwater lakes, including the northern shore of Lake Superior. Preferred breeding lakes are clear and oligotrophic, surrounded by forest, with rocky shorelines, deeply indented bays, numerous islands, and floating bogs; such lakes are characteristic of boreal and mixed forest overlying Precambrian shield (Barr 1973, 1996).

Common loons feed opportunistically on available species of live fish, including (but not limited to) lake whitefish, lake trout, and longnose sucker (McIntyre and Barr 1997). Most fish taken by common loons are 10 grams (g) to 250 g (Barr 1986). In addition to fish, common loons may consume other aquatic vertebrates, some invertebrates, and occasional vegetation (Barr 1996). Crustaceans (e.g., crayfish [Decapoda]) constitute a major part of the loon diet when fish are scarce or water is murky (1.0 m visibility) (Barr 1973). Loons primarily feed along shorelines with good underwater visibility and low density vegetation. While feeding is usually concentrated in the upper 5 m of the water column, common loons may dive deeper (to 60 m) in clear water (Roberts 1932).

While the normalized daily food ingestion rate (FIR) for chicks is 0.40 grams per gram per day (g/g-day) at week 1, it gradually declines to 0.22 g/g-day by fledging (11 weeks after hatching). The normalized FIR remains approximately constant (averaging 900 grams per day or g/day) into second year, but varies with environmental conditions and activity (McIntyre and Barr 1997).

Common loons are territorial, with an average territory size in Ontario of 70.4 ha (range 7 ha to 200 ha, n = 420; Barr 1973). Thus, JC might support a single pair of common loons, while RPH might support as many as 15 pairs. In addition, one or two pairs may occupy territories that include portions of both JC and RPH. The 2005 North American Breeding Bird Survey for the Marathon, Ontario survey route (http://www.mbr-pwrc.usgs.gov/cgi-bin/rtena25.pl?68078) reported 1.33 common loons/route.3

2.2.3.2 Bald Eagles

The bald eagle is a member of the Falconiform order and is often seen flying over water, hovering, and then plunging feet first to catch fish in its talons. As is typical of birds of prey, the bald eagle is large and has a sharp hooked bill and powerful talons. Body weights of bald eagles vary across subspecies and geographic regions, but are among the largest birds of prey in North America (3.0 kilograms (kg) to more than 7.0 kg). A body weight of 5.4 kg is assumed in this ERA based on the mean of body weights reported by USEPA (2004).

The bald eagle's breeding range includes limited areas of southern Ontario (Cadman et al. 1987, as cited in Buehler 2000), as well as other extensive portions of North America. Its wintering range is generally south of its breeding range, primarily along estuaries and rivers in southern Canada and larger water bodies in the United States. Bald eagles can be found in a broad array of habitats, including coast, beach, shore, river, stream, and riparian. However, common habitat attributes include an adequate supply of fish, shallow waters (preferred foraging for fish accessibility), elevated nest and perching sites (e.g., trees, bluffs), and an ice-free breeding season.

Studies cited by USEPA (2004) on the bald eagles' dietary composition typically show a diet dominated by fish (between 70% and 90%). Fish captured by bald eagles generally measure about 25 cm to 35 cm in length (Haywood and Ohmart 1986). Given their foraging strategy (i.e., diving feet first and only accessing the top 1 m of water), prey are limited to surface-schooling fish and those inhabiting shallows. Craig et al. (1988) reports a range of FIR for free-flying adult eagles in Connecticut, USA from 0.12 g/g-day to 0.14 g/g-day.

Bald eagles generally have large home ranges, averaging from approximately 1,000 ha to 2,800 ha. Nesting eagles typically forage within 0.5 km of the nest but they can travel up to 3 km to 8 km from the nest to obtain food (USEPA 2004). Nest densities are strongly influenced by habitat quality (e.g., abundance of large nesting trees) and the type of water body, but nest densities ranging from 0.03 nests per km to 0.4 nests per km of shoreline have been reported (USEPA 1993). Thus, depending upon the availability of nesting platforms and prey, JC and RPH could theoretically support several breeding pairs. Although the AOC lies within the bald

[3] Non-integer value is attributable to averaging of records across multiple observers.

eagle's range, this species was not among those observed during the 2005 North American Breeding Bird Survey for the Marathon, Ontario survey route (http://www.mbr-pwrc.usgs.gov/cgi-bin/rtena25.pl?68078). However, bald eagles are expected to forage occasionally in Peninsula Harbour.

2.2.4 Piscivorous Mammal Populations

As previously noted, the VEC of piscivorous mammal populations is represented by mink. Although other mammals, such as raccoons, fishers, and martens may forage along the shoreline of Peninsula Harbour, mink are expected to be the most highly exposed and most sensitive mammal in the region. They are also the most abundant and widespread carnivorous mammal in North America (USEPA 1993). Thus, mink serve as a conservative surrogate for other mammalian species inhabiting the region. The following life history information is largely excerpted from USEPA (1993).

Body sizes of mink vary greatly throughout the species' range, with males weighing as much as twice females in some populations. Based on multiple studies presented by USEPA (1993), body weights range from 0.44 kg to 0.93 kg and average 0.69 kg.

Mink are associated with aquatic habitats of all kinds, including rivers, streams, lakes, swamps, marshes, and back water areas (Linscombe et al. 1982). Mink prefer irregular shorelines to more open, exposed banks (Allen 1986). They also tend to use brushy or wooded cover adjacent to the water, where cover for prey is abundant and where downfall and debris provide den sites (Allen 1986).

Mink are opportunistic feeders, taking whatever prey is abundant (Hamilton 1936, 1940, Errington 1954, Sargeant et al. 1973). The most important prey for mink year-round are mammals (Eagle and Whitman 1987), but they also hunt aquatic organisms and other terrestrial prey, depending on the season (Linscombe et al. 1982). It is assumed that the diet of mink living near Peninsula Harbour is equally divided between aquatic and terrestrial prey (Appendix D).

The home range of mink encompasses both their foraging areas around waterways and their dens. The shape of mink home ranges depends on habitat type; riverine home ranges are basically linear, whereas those in marsh habitats tend to be more circular (Birks and Linn 1982; Eagle and Whitman 1987). Home range size depends mostly on food abundance, but also on the age and sex of the mink, season, and social stability (Arnold 1986, Birks and Linn 1982, Eagle and Whitman 1987, Linn and Birks 1981, Mitchell 1961). Gerell (1970) reported that home ranges of adult mink range from 1.0 km to 5.0 km of shoreline. Thus, JC could theoretically support one mink pair, while RPH could theoretically support about 5 to 20 pairs.

The quality of available habitat for mink and river otter along the shore of Peninsula Harbour was evaluated in 2007, (Appendix D). A detailed habitat survey, performed on foot, recorded evidence of mink or otter, including tracks, scat, potential or occupied den locations, otter slides, and otter latrines. Several indications of the presence of mink and river otter were observed and a mink may have been observed once, although species identification could not be confirmed. Field data were incorporated into a model, developed by the U.S. Fish and Wildlife Service (USFWS), to determine the suitability of habitat at a given location relative to the habitat requirements of mink. Both the model and survey information show that constraints on habitat suitability within the Peninsula Harbour shoreline, primarily related to limited vegetative cover near the shoreline indicate that population density is likely lower than would be suggested based on home range areas alone.

2.3 Assessment and Measurement Endpoints

Assessment endpoints are explicit expressions of the environmental values that are to be protected, operationally defined by an ecological entity (e.g., fish, birds, mammals) and its attributes (e.g., community structure, survival, growth, reproduction). Assessment endpoints are selected based on ecological relevance, susceptibility, and relevance to management goals.

Based on the CSM and VECs, the following assessment endpoints are selected for evaluation in this ERA: 1) community structure, survival, and reproduction of benthic invertebrates; 2) survival and reproduction of fish; 3) survival and reproduction of piscivorous birds; and 4) survival and reproduction of piscivorous mammals.

A measurement endpoint is defined as a measurable ecological characteristic that is related to the valued characteristic chosen as the assessment endpoint and is a measure of biological effects. In some cases, it is possible to directly measure the assessment endpoints selected for evaluation (e.g., surveys of biological community quality). Direct measurement of assessment endpoints minimizes the need to extrapolate between the measurement and the goal. Comparisons of estimated exposures with toxicological information for each COC facilitate the interpretation of biological community data and serve as the primary measurement endpoint where biological community data are not available. Thus, more than one measurement endpoint may be selected for a given assessment endpoint.

Measurement endpoints selected to evaluate potential risks to benthic invertebrates are: 1) concentrations of mercury and PCBs in sediment in relation to appropriate sediment quality guidelines and concentrations reported in literature to be harmful to benthos; 2) sediment toxicity to multiple invertebrate species, as measured in laboratory toxicity tests; and 3) abundance, richness, and diversity of assemblages, relative to reference stations of comparable habitat characteristics. These measurement endpoints were investigated by Milani et al. (2002) as summarized in Section 4.3 of this ERA.

Measurement endpoints selected to evaluate potential risks to fish are: 1) concentrations of mercury and PCBs in tissues of representative species in relation to concentrations reported in literature to be harmful to fish; 2) fish community structure and recruitment relative to reference area(s); and 3) comparison of average total mercury and PCB concentrations in extrapolated whole body longnose sucker, lake trout, and lake whitefish samples from the AOC to concentrations in the same species from a local Lake Superior reference area (Zone 7 of the Great Lakes SFCMP.

Measurement endpoints for evaluating risks to piscivorous wildlife are: 1) comparison of modeled dietary intake of COCs by two representative avian species (common loons and bald eagles) and one representative mammalian species (mink) to doses reported in the literature as thresholds for adverse effects on survival or reproduction (i.e., hazard quotients or HQs); and 2) comparison of species-specific and location-specific whole body fish and invertebrate tissue concentrations of mercury and PCBs to Canadian tissue residue guidelines (TRGs) for the protection of wildlife that consume aquatic biota.

The next three sections of the ERA, Sections 3 through 5, present the methods and results of the evaluation of the ecological measurement endpoints.

3 ECOLOGICAL EXPOSURE ASSESSMENT

Exposure assessment is the process of measuring or estimating the magnitude, frequency, and duration of wildlife exposures to chemicals present in the environment (USEPA 1992a). An ecological exposure assessment builds on the qualitative descriptions in the CSM to quantitatively estimate the exposure of VECs to mercury and PCBs. Concentrations of mercury and PCBs in sediment provide the basis for assessing the relationship between exposures and observed effects on benthic invertebrates. Exposure of fish is quantified as mercury and PCB concentrations in fish tissue. Exposures of piscivorous birds and mammals are estimated in three ways: 1) using the equation for dietary intake by wildlife provided in USEPA's (1993) Wildlife Exposure Factors Handbook, for use in generating HQs; 2) estimating PCB body burdens in mink using methods developed by Fuchsman et al. (2008); and 3) comparing fish species-specific and location-specific concentrations of mercury and PCBs in fish tissue to TRGs. These practices are further described below.

3.1 Benthic Invertebrate Exposure

Milani and Grapentine (2002) directly exposed invertebrates to site sediment in a laboratory setting, to evaluate potential sediment toxicity, in accordance with BEAST methodology (Reynoldson et al. 1995, 2000). These authors' assessment of benthic invertebrate community quality also directly integrates exposure and effects of chemicals in sediment, as well as effects of physical habitat variables. The BEAST data set (Milani and Grapentine 2002) included 21 stations in JC, 8 stations in RPH, and 4 stations outside of Peninsula Harbour. The JC sediment contained up to 19.5 mg/kg total mercury. By comparison, more than 90% of all other surface sediment samples collected from JC (i.e., during other sampling events) contained mercury concentrations below this concentration. Thus, the BEAST data set provides a good representation of the range of sediment mercury concentrations to which benthic invertebrates in JC are exposed. The relationship between site-specific biological effects and sediment COC concentrations is discussed in Section 5. Concentrations of mercury and PCBs in invertebrate tissue are summarized in Tables 3-1 and 3-2, respectively.

In considering sediment chemistry data relative to potential effects on benthic invertebrates, the mean and 95th percentile concentrations provide a measure of exposure. The percentage of samples exceeding various sediment screening values is also considered. Total mercury concentrations in JC sediment ranged from non-detect (<0.03 mg/kg) to 51 mg/kg, with a mean of 6.4 mg/kg and a 95th percentile of 23 mg/kg. Total mercury concentrations in RPH sediment ranged from non-detect (<0.03 mg/kg) to 2.3 mg/kg, with a mean of 0.73 mg/kg and a 95th percentile of 1.9 mg/kg.

Concentrations of total PCBs in sediment are considered on a dry weight basis (as mg/kg) or an organic carbon normalized basis (as micrograms PCB per gram organic carbon; µg/goc), depending on the toxicity value used for comparison. As an example, a concentration of 1 mg/kg in sediment containing 2.5% organic carbon is equivalent to 40 µg/goc. On a dry weight basis, total PCB concentrations in JC sediment ranged from non-detect (<0.02 mg/kg) to 1.1 mg/kg, with a mean of 0.21 mg/kg and a 95th percentile of 0.64 mg/kg. On an organic carbon basis, the mean and 95th percentile concentrations in JC were 17 and 47 µg/goc, respectively. In RPH, dry weight total PCB concentrations in sediment ranged from non-detect (<0.04 mg/kg) to 0.18 mg/kg, with a mean of 0.08 mg/kg and a 95th percentile of 0.18 mg/kg. The organic carbon normalized mean and 95th percentile PCB concentrations in RPH were 7.8 and 20 µg/goc, respectively. Concentrations of total PCBs in surface sediment were not significantly different between JC and RPH, based on the Wilcoxon rank sum test ($p > 0.05$).

3.2 Fish Exposure

Exposures of fish to mercury and PCBs were evaluated in this ERA based on concentrations in fish tissue. Fish are exposed to these bioaccumulative COCs through ingestion of prey, direct contact with surface water, and contact with and incidental ingestion of sediment. Because mercury and PCBs are not metabolized significantly in fish, tissue concentrations integrate all of these exposure pathways and serve as the most direct and appropriate measure of exposure. As discussed in Section 1.3, in the absence of information on the forms of mercury present in fish tissue, this ERA conservatively assumes that all mercury detected in fish tissue is present as methylmercury.

For comparability with available effects data, measured and estimated whole body concentrations (see Section 1.3) are used to assess fish exposures. Mean and 95th percentile concentrations are used to estimate exposures to fish. Concentrations of mercury and PCBs in whole fish are summarized in Tables 3-1 and 3-2, respectively.

Whereas measured fish tissue concentrations are used to assess risks under current conditions, modeled concentrations are used in Section 7 to predict future risks under various remediation scenarios. Future risks are calculated for longnose sucker, because this species contains the highest mercury concentrations among the species sampled. These calculations require an assumption regarding longnose suckers' relative use of JC and RPH. Longnose suckers prefer water depths less than 37 m (Becker 1983), the maximum depth noted throughout the AOC. They exhibit both daily and seasonal movements, tending to move offshore during daylight and in the fall, and into streams during spring spawning (Becker 1983; Carlander 1969). Although JC makes up only 10% of the AOC, aquatic vegetation in JC may be somewhat attractive, particularly for younger fish. However, in the absence of a mark-and-recapture study or radio tagging study, the movements of longnose suckers between RPH and JC are uncertain. Purely on the basis of the relative areas of JC and RPH, as well as the greater shelter and vegetation assumed present in JC, the scenario in which longnose suckers derive 25% of their exposure from JC and 75% from RPH is plausible. Nonetheless, several alternative scenarios are evaluated in Section 7.2.2.1.

3.3 Wildlife Exposure

As noted above, exposures of wildlife (piscivorous birds and mammals) to mercury and PCBs are characterized in three ways in this ERA. Dose-based exposure characterization allows comparison of estimated daily intakes (DIs) of mercury and PCBs to dose-based toxicity reference values (TRVs) (in units of milligrams per kilogram body weight per day or mg/kg-day), while diet-based exposure characterization allows comparison of measured or extrapolated whole body fish and invertebrate tissue concentrations to TRGs (Environment Canada 1998, 2002) (in units of milligrams per kilogram of fish or mg/kg). Additionally, as a supplemental line of evidence in assessing PCB-related risks to mink, PCB body burdens in mink are estimated according to methods developed by Fuchsman et al. (2008). These approaches to characterizing wildlife exposure are described below.

3.3.1 Dose-Based Exposure to Wildlife

The dose-based approach for modeling exposure to wildlife VECs (i.e., common loons, bald eagles, mink) employs an algorithm for calculating DI of COCs via ingestion of prey:

Eqn. 2

$$DI = \sum_{i=1}^{n} (Ci \times Pi \times FIR) \times 1/BW$$

Where:

DI	=	daily intake (mg/kg-day)
Ci	=	concentration in ith dietary item (mg/kg)
Pi	=	fraction of diet as item i (unitless)
FIR	=	food ingestion rate kilograms per day or kg/day
BW	=	body weight (kg)

Calculated DIs for all wildlife receptors and all scenarios are summarized in Table 3-3.

Dietary concentrations (Ci) were generated in four steps. First, it was assumed that wildlife receptors consume various species of fish in the same proportions that they are represented in the sample database for JC and RPH. This practice assumes that any preferential sampling by species that occurred during fish collection efforts is generally consistent with prey consumption patterns exhibited by the wildlife receptors.

Second, because most of the fish samples collected were considerably larger than the representative wildlife receptors would normally consume, concentrations that would be expected to be present in prey of the size actually consumed were extrapolated from existing data using exponential regression analysis, as detailed in Appendix C. As described in Appendix C, the exponential regression formula was then used to estimate fish-specific concentrations scaled to the median of the preferred prey size range for each wildlife receptor.

Third, the size-normalized concentrations in samples of species expected to be consumed by each representative wildlife receptor were then grouped and arithmetic mean and 95% upper confidence limit of the mean (95% UCLM) concentrations were calculated separately for JC and RPH. The 95% UCLM concentrations were calculated using Hall's bootstrap method (Hall 1992), as described in Appendix C. Bootstrapping is the preferred method of calculating the 95% UCLM in this case because the dataset was divided between JC and RPH, which potentially truncates distributions. Many alternative methods of calculating 95% UCLMs assume a particular—untruncated—distribution to the dataset and thus would be invalid for this application. In contrast, bootstrapping generally assumes no particular distribution. Hall's Bootstrap method, however, is designed for use with skewed datasets, which are common for environmental data. This approach is more reliable than USEPA's confidence limit calculating software (i.e., ProUCL) for the reasons discussed above, as well as because ProUCL periodically malfunctions. Furthermore, it usually recommends Chebyshev's inequality based upper confidence limit, which yields a tolerance limit, rather than an upper confidence limit.

Fourth, AUFs were used to adjust the JC- and RPH-specific concentrations according to five varying proportions of the wildlife diet assumed to be derived from JC and RPH. These AUFs help account for the fact that there is no physical barrier to movement of fish or wildlife between JC and RPH. Scenario 1 assumes all prey was derived from JC. Scenario 2 assumes that 75% of prey is derived from JC and 25% from RPH. Scenario 3 assumes that 50% of prey is derived from JC and 50% from RPH. Scenario 4 assumes that 25% of prey is derived from JC and 75% from RPH. Scenario 5 assumes that 100% of prey is derived from RPH. Simply based on the relative areas and length of shoreline of JC and RPH, Scenario 4 appears most realistic, yet appropriately conservative. However, unique preferences of different fish species and habitat characteristics of JC and RPH may render other scenarios plausible.

Finally, in the absence of information on the forms of mercury present in AOC fish tissue, this ERA makes the conservative assumption that all mercury detected in fish tissue is present as methylmercury. Tables 3-4 and 3-5 summarize the dietary concentrations employed for each wildlife receptor and for each scenario, for mercury and PCBs, respectively. It is worth noting that the estimated dietary concentrations of methylmercury in aquatic invertebrates shown in Table 3-4 indicate slightly higher concentrations in RPH (represented by Scenario 5) than in JC (represented by Scenario 1). This result is unexpected, given the higher concentrations of methylmercury in JC sediment, compared to RPH sediment. This finding is likely an artefact of the small sample size (n=2) for aquatic invertebrates from RPH that were analyzed for methylmercury. It appears that the two invertebrate samples collected from RPH and analyzed for methylmercury contained elevated concentrations of methylmercury relative to JC. A larger sample size likely would have yielded more representative results. However, given that aquatic invertebrates constitute a relatively minor portion of the diet of mink and common loon (and none of the diet of the bald eagle), this potential limitation to the data is unlikely to significantly affect the outcome of the ERA.

Empirical data on FIR are available for few wildlife species, primarily due to the difficulty of measuring feeding rates for free-ranging wildlife. Measured FIRs for captive animals are not used in this ERA because such animals do not expend energy foraging for food and water, avoiding predators, defending territories, etc. (USEPA 1993). Therefore, in the absence of empirical data on FIR in free-ranging animals, FIR is calculated in this ERA from allometric equations developed from the free metabolic rate (FMR) of free-ranging animals. FIR is derived from FMR as follows:

Eqn. 3

$$FIR = \frac{FMR \times CF}{\sum_{i=1}^{n} AEi \times Gi \times Pi}$$

Where:

FIR	=	food ingestion rate (kg/day)
FMR	=	free metabolic rate (kiloJoule per day or kJ/day)
CF	=	conversion factor (0.239 kcal/kJ)
AEi	=	assimilation efficiency of ith dietary item (unitless)
Gi	=	gross energy of ith dietary item (kcal/kg)
Pi	=	fraction of diet as item i (unitless)

FMR is calculated from Nagy (1987):

Eqn. 4

$$\mathrm{FMR} = \mathrm{a} \times \mathrm{BW}^{\mathrm{b}}$$

Where:

FMR	=	free metabolic rate (kJ/day)
a	=	slope (unitless)
BW	=	body weight (kg)
b	=	power (unitless)

Input variables used in these equations are detailed in Tables 3-6 through 3-8 for common loons, bald eagles, and mink, respectively. For all wildlife VECs, the inputs required for the calculation of the FIR (i.e., FMR, gross energy, and assimilation efficiency) are drawn from USEPA (1993). Because the selection of these values closely followed USEPA (1993) guidance, the rationale for their selection is not discussed further.

3.3.1.1 Common Loon

Assumptions regarding prey preferences and body weights for common loons were drawn from McIntyre and Barr (1997) and sources cited by them. In considering these secondary sources, appropriate data were selected with preference given to data on adult loons inhabiting Canada (particularly Ontario). Exposure parameter values for common loons that are applied in this ERA are summarized in Table 3-6. Based on Barr (1996), this ERA assumes that 20% of the diet of common loons is comprised of aquatic invertebrates, while the remaining 80% is comprised of fish ranging in mass from 10 g to 250 g (Barr 1986), which is equivalent to 7 cm to 31 cm, based on the length-weight conversions provided on www.FishBase.org. An average body weight of 4.0 kg is applied in this ERA, based on Barr's (1986) observation that male common loons from Northwestern Ontario averaged 4.4 kg (n = 23), while females averaged 3.54 kg (n = 15).

3.3.1.2 Bald Eagles

Assumptions regarding prey preferences and body weights for bald eagles are drawn from USEPA (1993, 2004). USEPA (1993) is the Wildlife Exposure Factors Handbook, while USEPA (2004) is a publicly available and peer-reviewed comprehensive probabilistic ERA for the Housatonic Rest of River (Massachusetts, USA) site. Exposure parameter values for bald eagles that are applied in this ERA are summarized in Table 3-7. The assumed body weight for bald eagles is based on the meta-analysis of multiple studies presented by USEPA (2004). The body weight employed in this ERA is set equal to the mean of USEPA's (2004) normal distribution for bald eagle body weights. The various studies cited by USEPA (2004) on dietary composition show a diet primarily dominated by fish (71% to 90%). The point estimate value (0.76) employed by USEPA (2004) is used in this ERA. Fish captured by bald eagles generally measure about 25 cm to 35 cm in length (Haywood and Ohmart 1986). The remainder of the bald eagle's diet is generally made up of mammals. In the absence of any data on concentrations of mercury and PCBs in mammals, that portion of the bald eagle's diet is assumed to be uncontaminated. This is a reasonable assumption because bald eagles typically forage in undeveloped areas; such undeveloped areas are not expected to be impacted by mercury or PCBs.

3.3.1.3 Mink

Assumptions regarding prey preferences and body weights for mink are drawn from USEPA (1993, 2004). Exposure parameter values for mink that are applied in this ERA are summarized in Table 3-8. Based on the life history characteristics presented above, for purposes of this ERA, it is assumed that resident mink consume a diet composed of equal portions of terrestrial and aquatic prey. In the absence of any data on concentrations of mercury and PCBs in terrestrial prey, that portion of the mink's diet is assumed to be uncontaminated. This is a reasonable assumption because mink typically forage in undeveloped areas; such undeveloped areas are not expected to be impacted by mercury and PCBs. The assumption that half of the mink's diet is aquatic is conservative, in that some studies (McDonnell and Gilbert 1981, Proulx et al. 1987, Cowan and Reilly 1973) indicate that as little as 20% of the mink's diet may be aquatic. Of the aquatic prey, 96% is assumed to be composed of fish ranging in length from 4 cm to 25 cm and 4% is assumed to be composed of aquatic invertebrates. It is assumed that the fish portion of the mink's diet is comprised of equal proportions of lake trout, lake whitefish, and longnose sucker within the stated size range. The body weight of 0.69 kg used in this ERA is based on USEPA's (2004) meta-analysis of body weight data from multiple studies.

3.3.2 Mink Exposure Based on Body Burden

In addition to estimating mink exposures based on the dose of total PCBs, mink body burdens are estimated in Table 3-9. This approach accounts for the bioaccumulation potential of site-specific PCB mixtures, in order to decrease uncertainty in dose-response estimates. Methods used to estimate and interpret mink body burdens are based on a recent meta-analysis of PCB effects on mink (Fuchsman et al. 2008).

As reviewed by Fuchsman et al. (2008), several options are available for quantifying mink exposures to PCBs. Some options yield much closer relationships than others with observed adverse effects on mink. The most effective exposure metric is the estimation of total PCB concentrations in mink, based on known concentrations of PCB homologues (e.g., total tetrachlorobiphenyls, total pentachlorobiphenyls) in the mink's diet. The second-most effective exposure metric is the estimation of TEQ concentrations in mink, based on known concentrations of PCB congeners in the diet and toxicity equivalence factors (TEFs) recommended by USEPA (2003) for assessing internal doses. (Note that the use of these alternative TEFs is considerably more effective than the 2005 World Health Organization TEFs for predicting adverse effects on mink.) Because fish collected in 2007 were analyzed for PCB homologues and congeners, both of these methods are considered as lines of evidence in the ERA for mink.

Total TEQ body burdens in mink are estimated as:

Eqn. 5

$$C = TEF\frac{AD}{K}\left(1 - e^{-Kt}\right)$$

Where:

C = whole body concentration (µg/kg)
TEF = toxicity equivalence factor
A = assimilation efficiency (fraction)
D = daily intake (µg/kg-day)
K = elimination rate (fraction/day)
t = exposure duration (days)

Total PCB body burdens in mink are estimated from homologue concentrations using the same equation, except that the TEF term is omitted. Homologue- and congener-specific values for TEF, A, and K are as listed by Fuchsman et al. (2008). Exposure duration is assumed to equal three years (1,095 days), which is the typical lifespan of mink in the wild (Lariviere 1999).

In order to identify homologue- and congener-specific daily intake rates, a site-specific congener fingerprint was identified from concentrations in 24 fish tissue samples analyzed for total PCBs, homologues, and congeners. Specifically, the concentration of each homologue or congener was normalized based on the total PCB concentration in the same sample. The resulting relative homologue and congener concentrations (µg homologue or congener per mg total PCB) were multiplied by the total PCB doses calculated for mink (Section 3.3.1), thus taking into account the AUFs and dietary preferences identified for mink in this ERA. Calculated homologue body burdens were then summed to estimate total PCB concentrations in mink, and congener TEQ body burdens were summed to estimate total TEQ in mink.

3.3.3 Wildlife Exposure Based on Dietary Concentration

In addition to the dose-based and body burden-based exposure metrics described above, exposure to wildlife is also evaluated in this risk assessment based on direct evaluation of prey tissue concentrations. As illustrated in Tables 3-10 and 3-11, concentrations of mercury and PCBs in whole body invertebrate and fish tissue are compared to TRGs (Environment Canada 2002), in order to understand the proportion of invertebrate and fish samples that exceed those guidelines. In addition to comparing the mean and 95% UCLM concentrations to these guidelines, the percentage of samples (of each species) exceeding the guidelines is provided. The basis for the TRGs is discussed in Section 4.5.6 of the ecological hazard assessment, while the outcome of the comparison is discussed in Section 5.3 of the ecological risk characterization.

4 ECOLOGICAL HAZARD ASSESSMENT

The ecological hazard assessment evaluates the potential for mercury and PCBs to cause adverse effects in exposed VECs and estimates the relationship between the extent of exposure and the severity of effects. This section opens with an overview of the chemical, physical and toxicological characteristics of mercury and PCBs, before presenting hazard assessments specific to each VEC group. For those measurement endpoints that are based on direct observation of effects (i.e., benthic invertebrate toxicity and community structure, fish community structure and recruitment), the hazard assessment analyzes of site-specific biological data to determine differences relative to reference areas and/or correlations with mercury and PCB concentrations. For all other measurement endpoints, the hazard assessment reviews the pertinent literature and selects the TRVs that are used to interpret the potential for adverse effects. For benthic invertebrates and fish, TRVs are literature-based concentrations in sediment or tissue (i.e., in units of mass per mass, such as mg/kg), below which adverse effects are unlikely. For piscivorous birds and mammals, TRVs include both literature-based doses (i.e., in units of mg/kg-day) and TRGs (i.e., in units of mg/kg), below which adverse effects are unlikely.

4.1 Overview of Mercury

Mercury is the only metal that is liquid at room temperature. It has a high surface tension, forming spherical droplets when the liquid is released. It has low solubility in water and is volatile at ambient temperature (Environment Canada 2002). The element has two principal valence (Hg^0 and Hg^{2+}) states and is found in the environment in the metallic form and in the form of various inorganic and organic complexes. The specific state and form in which the compound is found in an environmental medium depends on a number of factors, including the reduction-oxidation (redox) potential and pH of the medium (ATSDR 1999).

As described by ATSDR (1999), mercury is transformed in the environment by biotic and abiotic oxidation and reduction, bioconversion of inorganic and organic forms, and photodegradation of organomercurials. Inorganic mercury can be methylated by microorganisms indigenous to soil, sediment, fresh water, and salt water. Although this process is mediated by various microbial populations under both aerobic and anaerobic conditions, sulphate-reducing bacteria are responsible for most methylation in the environment (Gilmour and Henry 1991), with anaerobic conditions favouring their activity (Regnell and Tunlid 1991). The methylation of inorganic mercury by sulphate-reducing bacteria is enzymatically catalyzed and involves multiple possible metabolic pathways. The rate of methylmercury formation by these pathways is influenced by factors including enzyme availability, microbial metabolic rate, substrate quality, and factors that influence the rate at which inorganic Hg^{2+} enters the cell (the balance between methylation vs. demethylation potential).

Although inorganic mercury is the dominant form in the environment and is easily accumulated, it is also more quickly depurated. As detailed by Nichols et al. (1999), methylmercury in the diet is absorbed with high efficiency in the vertebrate digestive tract and associates rapidly with sulfhydryl-containing molecules in blood. These mobile complexes transport methylmercury to tissues and organs, and facilitate its movement across cell membranes. Thus, because methylmercury accumulates quickly and is depurated very slowly (Clarkson 1994), it has a greater potential to biomagnify in higher-trophic-level species. The fraction of total mercury existing as methylmercury typically increases with trophic level (i.e., from primary producers to fish to piscivorous birds and mammals). Nearly all (95% to 100%) of the mercury present in fish is methylmercury, obtained mostly from the diet (Grieb et al. 1990; Bloom 1992). The half-life of total mercury in fish is approximately five days to five months, while the half-life of methylmercury in fish is from one to three years (USEPA 1997c). While the ecotoxicological impacts of mercury have been well recognized since the 1950s, records of its potential as a toxicant date back to the early-1860s (Watanabe and Satoh 1996). Mercury is easily transported across cell membranes, resulting in toxicity to biota.

4.2 Overview of PCBs

As largely excerpted from National Research Council (NRC) (2001), PCBs are entirely anthropogenic in origin and were manufactured in the United States between 1929 and 1977, with their heaviest industrial use occurring in the 1950s and 1960s. PCBs were never manufactured in Canada (www.ec.gc.ca). The structure of PCBs (i.e., two hexagonal rings of carbon atoms connected by single bonds) is highly stable. PCBs consist of 209 possible chemical structures (known as congeners), defined by the number and position of chlorine atoms on the biphenyl molecule. Between one and ten chlorine atoms have the potential to substitute for hydrogen atoms on the biphenyl rings. PCBs are subdivided into groups called homologues based on the number of chlorine atoms per biphenyl molecule. Typically, industrial PCBs were sold in complex mixtures (known as Aroclors) composed of many (50 to 60) congeners and classified by percentage of chlorine.

Because the toxicity of PCBs varies across congeners, characterization of the toxicity of mixtures of PCBs is complex. Many of the effects of certain congeners ("dioxin-like" PCBs) are mediated through the interaction with the arylhydrocarbon receptor (AhR), and are consequently similar to the effects of dioxin. The strength with which individual PCB congeners bind to the AhR is correlated to their ability to elicit dioxin-like effects, leading to the concept of 2,3,7,8-TCDD TEQs. TEFs are used to express the toxicity of a given PCB congener relative to 2,3,7,8-TCDD, the most potent form of dioxin. The products of each congener's concentration and its TEF are summed to yield a TEQ concentration for any mixture of dioxin-like congeners.

One of the main endpoints upon which TEFs are based is the induction of CYP1A1. The binding of the PCB molecule with the AhR receptor initiates a series of cellular events, leading to the transcription of the gene corresponding to CYP1A1, and ultimately, enhanced synthesis of the CYP1A1 enzyme. Typically, the congeners with the highest TEF values are planar and have high degrees of chlorine substitution. Noncoplanar congeners and those with low chlorine substitution have low TEF values, but have been associated with immunological and neurobehavioral endpoints. Since the toxicity of noncoplanar PCBs is not mediated by the AhR and not accounted for in the TEF approach, it is also important to consider non-dioxin-like risks.

Studies have shown that the neurotoxic effects of the noncoplanar PCBs are mediated by signal transduction pathways, rather than the AhR. PCB congeners have been shown to affect tyrosine kinase, protein kinase C, phospholipase A2, and intracellular calcium homeostasis. In addition, PCBs may interact with one or more steroid receptors, leading to estrogenic and antiestrogenic effects. Thyroid hormone metabolism and the immune system may also be affected by PCBs. An increase in oxidative stress due to PCBs may lead to carcinogenesis. The metabolism of PCBs has the potential to increase their toxicity as well. Because a relatively few fish tissue samples collected from the AOC were analyzed for congeners and because these samples had a relatively low concentration of total PCBs, quantitative evaluation of TEQ risks would be neither conservative nor representative. Thus, the focus of PCB hazard assessment in this ERA is based on total PCBs.

Bioaccumulation appears to be the main route of ecological exposure to PCBs. Since PCBs bind strongly to organic particles, aquatic organisms are exposed to a combination of dissolved, sediment-associated, and food-associated PCBs.

4.3 Hazard Assessment for Benthic Invertebrates

Milani et al. (2002) evaluated potential toxicity to benthic invertebrates in Peninsula Harbour using the BEAST methodology. This methodology includes multivariate analysis of benthic invertebrate community structure, laboratory toxicity testing, and chemical and physical characterization of sediment. Benthic community and toxicity test results are compared to biological criteria developed based on Great Lakes reference sites. Study results are summarized below, followed by a discussion of sediment quality screening values.

4.3.1 Community Structure Outcomes

Milani et al. (2002) evaluated 33 sampling stations, including 21 stations in JC, 8 stations in RPH, and 4 stations outside of Peninsula Harbour. The data were evaluated through standardized statistical comparisons to a database of references sites representative of Lake Superior, using BEAST software. This method classifies each sampling station into one of five possible Great Lakes reference groups, based on geographic location (latitude, longitude), sediment attributes (organic carbon, nitrogen, potassium, calcium, magnesium, manganese, silicon), and physical/chemical parameters (water depth, alkalinity, pH) (Environment Canada 2004). All 33 stations were classified as corresponding to Reference Group 5, which consists of 75 sites from Lake Superior (30), Georgian Bay (19), the North Channel (12), Lake Michigan (7), Lake Ontario (5), and Lake Huron (2) (Milani et al. 2002). Each of the 75 reference stations serves as a replicate for statistical comparison to benthic community characteristics at target stations.

The benthic community at JC stations was found to be consistently different than that of reference stations. The principal differences were higher diversity at JC sites and lower abundance of the amphipod *Diporeia hoyi* (formerly *Pontoporeia hoyi*). The latter finding at JC was attributed, at least in part, to a difference in water depth and sediment organic content between JC and the reference sites included in the BEAST model. *Diporeia hoyi* is characteristic of deep, oligotrophic sites (Reynoldson et al. 1995), whereas JC is less deep and is organically enriched due to past accumulation of woody material, particularly bark (Milani et al. 2002, Peninsula Harbour RAP Team 2002). With regard to water depth, the median depth at the JC stations was 12.5 m, while the median depth for the reference sites was 28.0 m. By comparison, *Diporeia* spp. are most common at depths greater than 30 m (Nalepa et al. 2005). Milani et al. (2002) did not customize the reference data set to be more closely comparable to JC in terms of water depth; therefore, it is not possible to distinguish the relative importance of water depth versus organic enrichment in explaining differences between site and reference conditions.

4.3.2 Toxicity Testing Outcomes

Sediment toxicity at the 33 sampling stations was evaluated using several subchronic and chronic tests, including:

- 28-day survival and growth of the amphipod (*Hyalella azteca)*;
- 10-day survival and growth of the midge (*Chironomus riparius)*;
- 21-day survival and growth of the mayfly (*Hexagenia spp.)*.; and
- 28-day survival and reproduction of the worm (*Tubifex tubifex)*.

Sediment from only 1 of 21 JC sampling stations was classified as toxic; the observed effects in this sample were attributed to physical characteristics (hard clay substrate). The mercury concentration in this sample was among the lowest measured; organic chemicals were not measured in this study. Three stations outside JC were considered toxic and one additional station outside JC was considered potentially toxic. Unusual physical characteristics were observed in two of these stations (very hard sand substrate or high percent clay). The cause of toxicity in the remaining two stations was not determined. All stations with toxicity exhibited low mercury concentrations relative to stations lacking toxicity.

4.3.3 Sediment Screening Values and Concentration-Response Studies

The MOE and the Canadian Council of Ministers of the Environment (CCME) have adopted sediment screening values for total mercury and total PCBs. In both cases, pairs of screening values are identified, with a lower value below which toxicity is unlikely (LEL and interim sediment quality guideline [ISQG], respectively), and an upper value above which toxicity is often observed [SEL] and probable effect level [PEL], respectively).

Sediment screening values for mercury are:

- LEL = 0.2 mg/kg
- SEL = 2 mg/kg
- ISQG = 0.17 mg/kg

- PEL = 0.486 mg/kg

Sediment screening values for PCBs are:

- LEL = 0.07 mg/kg
- SEL = 530 µg/goc
- ISQG = 0.034 mg/kg
- PEL = 0.277 mg/kg

These empirical screening values are developed using a co-occurrence approach, where data from biological monitoring at a large number of sites (e.g., information on the presence and absence of benthic organisms) are compared to the site chemistry data. It is widely accepted that empirical sediment quality guidelines, such as these, do not necessarily represent cause-effect, concentration-response relationships between chemical concentrations and biological effects (Wenning et al. 2005, Becker and Ginn 2008). Under the final Canada-Ontario Decision-Making Framework for Assessment of Great Lakes Contaminated Sediment (COA) (Environment Canada and Ontario Ministry of the Environment 2007) biological and toxicity studies are preferred over comparisons of sediment concentrations to guidelines, such as these.

Information that can be used to identify concentration-response relationships for chemicals in sediment includes spiked sediment studies, mechanistic approaches linking sediment porewater concentrations to toxicity, and toxicity data from sediment sites contaminated primarily with a single chemical of interest. Although such information is limited for mercury, Sferra et al. (1999) cite several examples of sites contaminated primarily with mercury, where toxicity was not observed at total mercury concentrations ranging from 5 mg/kg to 35 mg/kg, and even as high as 390 mg/kg in one case.

For PCBs, Fuchsman et al. (2006) reviewed multiple lines of evidence and identified a range of cause-effect screening values, depending on the homologue composition of PCBs in sediment. Although PCB homologues have not been measured in AOC sediment, the available congener data suggest that site-specific PCB mixtures are dominated by penta-, hexa-, and heptachlorobiphenyls, similar to the composition of Aroclor 1254. Fuchsman et al. (2006) identified a cause-effect sediment screening value for Aroclor 1254 of 1,500 µg/goc. For comparison, the lowest cause-effect screening value identified by Fuchsman et al. (2006) is 210 µg/goc (for Aroclor 1242, considered relatively bioavailable in sediment).

4.4 Hazard Assessment for Fish

The hazard assessment for fish includes site-specific biological information and published information relating tissue concentrations of COCs with adverse effects, as described below.

4.4.1 Community Structure Outcomes

Twenty years ago, Hamilton (1987) determined that fish community quality in Peninsula Harbour was relatively poor. The more recent fish community survey by BEAK (2000, 2001) focused on documenting fisheries resources in JC and Carden Cove, rather than on detecting toxicity-related impacts, if any. Survey methods and results are described in Section 2.2.2. The most abundant species (more than 20 individuals captured) in JC were slimy sculpin, longnose sucker, mottled sculpin, and round whitefish. In Carden Cove, the most abundant species were longnose sucker, lake chub, and round whitefish.

Although differences were noted between the two coves (i.e., fish in JC were more diverse but less abundant), differences in both physical habitat and chemical exposures may have contributed to the observed differences in the fish communities. Also, much of the difference in fish abundance was attributable to the capture of a large number of juvenile longnose sucker in Carden Cove; several other species were more abundant in JC. One of the most abundant species in JC, mottled sculpin, is generally considered to be intolerant of pollution (USEPA 1999). BEAK (2000) also qualitatively noted evidence of successful reproduction (i.e., presence of adult, juvenile, and young-of-year fish) in both coves.

These results are not indicative of severe adverse effects on fish survival or reproduction in JC. However, they do not conclusively demonstrate whether more subtle effects due to mercury and PCB exposures are or are not occurring.

4.4.2 Toxicity Reference Values for Mercury in Fish

Studies linking whole body mercury concentrations with chronic effects on fish, including reductions in reproductive success, growth, and survival, were identified from a review paper by Beckvar et al. (2005). All primary sources were obtained and reviewed to ensure the accuracy and relevance of the reported toxicity data. Study results applicable to mercury concentrations in adult whole fish are summarized in Table 4-1.

Beckvar et al. (2005) identified a concentration of 0.2 mg/kg as the most appropriate TRV for mercury in tissue of juvenile and adult fish. This concentration is equal to the no observed adverse effect level (NOAEL) identified from Matta et al. (2001), who evaluated mortality and reproduction of the mummichog (*Fundulus heteroclitus*). The lowest observed adverse effect level (LOAEL) noted in Table 4-1 is identified from Friedmann et al. (1996), who observed adverse effects on gonadal development in walleye containing 0.25 mg/kg mercury (the lowest concentration tested). Although this is only an indirect measure of potential reproductive effects, Hammerschmidt et al. (2002) demonstrated that in fathead minnows containing a similar tissue residue (0.39 mg/kg), impaired gonadal development was associated with impaired reproduction.

As an alternative to the TRV identified by Beckvar et al. (2005), one could consider the mercury concentration of 0.06 mg/kg measured in control fish by Friedmann et al. (1996) to be a NOAEL, and select a TRV between 0.06 and 0.25 mg/kg (e.g., the geometric mean). However, this approach results in a TRV that is similar to mercury concentrations in control fish from various other studies and thus is not plausible as toxicity threshold. Sensitivity to mercury varies significantly across fish species, as McKim et al. (1976) observed no reproductive toxicity in brook trout at concentrations as much as an order of magnitude higher than those identified as toxic by Matta et al. (2001) and Friedmann et al. (1996). Thus, Beckvar et al.'s (2005) recommended TRV of 0.20 mg/kg mercury in whole fish is adequately protective and appropriate for use in this ERA.

4.4.3 Toxicity Reference Values for PCBs in Fish

The toxicity of PCBs to fish (based on fish tissue concentrations) is characterized based on a review of the scientific literature. Relevant studies were identified primarily from recent compilations of data relating tissue concentrations of PCBs in fish and adverse effects on fish (Jarvinen and Ankley 1999, Monosson 1999/2000, Niimi 1996). All primary sources were obtained and reviewed. For this ERA, controlled laboratory studies were selected that report: 1) whole-body PCB concentrations in adult fish; and 2) reproductive success. Fish reproduction is the most sensitive endpoint for PCB-related effects (Jarvinen and Ankley 1999).

Studies reporting PCB concentrations in fry but not in adult fish were excluded from the toxicity characterization, for two reasons. First, PCB concentrations in fry are not directly comparable to concentrations in adult fish and thus are not comparable to the exposure data available for fish in Peninsula Harbour. Additionally, PCB concentrations in fry change rapidly with time, due to dilution of maternally transferred PCBs as the fry grow (e.g., Mac and Seelye 1981). As a result, the interpretation of fry PCB concentrations is confounded by fry age and degree of growth. Thus, total PCB concentrations in fry are not a reliable predictor of adverse effects on fish reproduction. Studies using individual PCB congeners (or mixtures of selected congeners) were also excluded, because the PCBs tested in such studies are not representative of environmentally relevant mixtures. Thus, the level of toxicity observed in such studies is quite likely to be very different than that associated with PCB mixtures occurring in the environment. Furthermore, sufficient data are available on environmentally relevant mixtures of PCBs to support a robust hazard assessment for fish, while quite limited congener-specific data are available for fish collected from the AOC.

Table 4-2 presents published data relating whole body PCB concentrations in adult fish to adverse effects on reproductive success. Of the test endpoints related to reproduction, larval survival and growth are more sensitive than other endpoints, such as fecundity (i.e., number of eggs produced), fertilization success, and hatching success. Differences in sensitivity are also apparent in the responses to different PCB formulations. For this ERA, the TRV is based on a study of reproductive success and larval survival in sheepshead minnows (Hansen et al. 1974). A TRV of 4.2 mg/kg is identified as the geometric mean of the NOAEL (1.9 mg/kg) and LOAEL (9.3 mg/kg) from this study.

4.5 Wildlife Hazard Assessment

The methodology used to derive the wildlife TRVs in this risk assessment is described below, followed by the rationale used in selecting TRVs for methylmercury and PCBs for both birds and mammals. The wildlife hazard assessment closes with a discussion of the TRGs (Environment Canada 1998, 2002) that are considered in the evaluation of risks to wildlife.

4.5.1 Methodology for Deriving TRVs

TRVs were derived based on the general methodology of Sample et al. (1996), by applying uncertainty factors (UFs) to laboratory study results, as detailed below:

Eqn. 6

$$TRV = \frac{Test\ Species\ Dose}{UF}$$

The test species dose is a daily dose of a chemical associated with a particular endpoint and effect. Test species doses for each COC-VEC pair (e.g., dose of methylmercury causing an adverse effect to mink, dose of PCBs causing an adverse effect to mink) were identified from appropriate literature references, with preference given to peer-reviewed primary sources. The following criteria are applied in selecting applicable studies:

- Relatedness of test species used in the study as compared to the wildlife species of interest – Studies on species that are similar with respect to taxonomic order and/or feeding guild are preferred over studies on species that are less closely related. In addition, studies on wild species are preferred over studies on domesticated species.
- Effects evaluated – Studies focused on most sensitive effects are preferred over studies on less sensitive effects; consequently, sublethal studies are preferred over lethal studies and studies on sensitive life stages are preferred over studies on adult non-breeding organisms.
- Type of endpoint – Studies with multiple dose groups that allow identification of a NOAEL and a LOAEL are preferred over studies that yield other endpoints or only a NOAEL or only a LOAEL.
- Duration of the dosing period – Lifetime or chronic duration studies are preferred over subchronic, acute, and single dose studies.
- Dose administration method – Studies utilizing dietary dosing are preferred over other oral dosing methods, which are preferred over injection, dermal, or inhalation dose administration.
- Chemical form tested – Studies utilizing methylmercury are preferred over those utilizing elemental or salt forms of mercury, while studies utilizing environmentally weathered mixtures of PCBs are preferred over those utilizing commercial mixtures or individual congeners.
- Documentation of study methods and quality control – Studies that clearly document the study design and methods that demonstrate adequate quality control are preferred over those that provide limited discussion on these topics.

NOAELs and LOAELs are commonly reported endpoints that may be considered in the selection of the test species dose. In most cases, this ERA bases the TRV on the geometric mean of the NOAEL and LOAEL from the most appropriate (i.e., critical) study. Because NOAELs and LOAELs are strongly influenced by the toxicity test study design, the true threshold of an effect is likely to fall between the two values. Thus, while the NOAEL may be an appropriate TRV for screening level ERAs, baseline ERAs such as this one strive to generate a more realistic estimate of risk. Consequently, the geometric mean of the NOAEL and LOAEL offers an appropriate basis for the TRV.

The geometric mean of the NOAEL and LOAEL used in TRV derivation is reported on – or converted to – a mg/kg-day basis. These units of dose allow comparisons among organisms of different body sizes (Sample et al. 1996). In cases where the critical study states the effect level or no effect level as a dietary concentration (i.e., in units of mg /kg food), the geometric mean of the effect level and no effect level is converted to a test species dose:

Eqn. 7

$$Dose = \frac{C \times IR}{BW}$$

Where:

Dose	=	test species dose of COC (mg/kg-day)
C	=	concentration of COC in food or water (mg/kg)
IR	=	ingestion rate of food or water by the test species (kg/day)
BW	=	body weight of the test species (kg)

UFs are typically identified based on three characteristics of the experimental conditions associated with the test species dose: 1) the duration of exposure; 2) the endpoint measured; and 3) differences in sensitivity among test and receptor species (Calabrese and Baldwin 1993, Ford et al. 1992, Opresko et al. 1994, Sample et al. 1996, USEPA 1996, Watkin and Stelljes 1993, Wentsel et al. 1994). As detailed below, the critical studies identified for methylmercury and PCB ecotoxicity employed either the same species as was selected as VECs or a closely related species. They were also chronic studies that yielded a NOAEL and/or a LOAEL. In some cases where only a NOAEL or LOAEL resulted from the critical study, it was assumed that the LOAEL is ten-fold higher than the NOAEL, for purposes of calculating the geometric mean of the NOAEL and LOAEL. Otherwise, it was not necessary to employ UFs in this ERA.

In addition to the approach used to identify TRVs, a dose response assessment was used as a supplemental line of evidence to determine effect concentrations for interpretation of mink body burdens of PCBs. Effect concentrations were identified from Fuchsman et al.'s (2008) evaluation of relationships between total PCB and TEQ concentrations in mink and reductions in overall reproductive success.

4.5.2 Avian TRVs for Methylmercury

Effects of methylmercury have been evaluated in both field and laboratory settings for a variety of avian species, including loons, egrets, quail, mallards, red-tailed hawks, zebra finches, and others. For purposes of deriving TRVs for loons and eagles in this ERA, we focused on those studies on the effects of methylmercury on piscivorous species that reported on ecologically pertinent endpoints (e.g., breeding and parenting behaviour, productivity) and yielded NOAELs and/or LOAELs. Consequently, less applicable studies—such as those conducted on species that are not piscivores and those conducted using other forms of mercury—were excluded from consideration in this effects characterization. For example, the two avian studies (Hill and Schaffner 1976, Heinz 1979) cited by Sample et al. (1996), as well as the mallard studies that formed the basis for USEPA's (1995) avian wildlife value for methylmercury were excluded from further consideration, given the availability of high quality studies on more relevant species.

Barr (1986) conducted a three-year field study on population dynamics (including breeding and parenting behaviour) of common loons breeding in northwestern Ontario and exposed to varying concentrations of methylmercury in fish, as well as to fluctuating water levels and turbidity. Barr (1986) found that behaviour and reproductive success of territorial loons were adversely affected in mercury-contaminated lakes downstream from a chlor-alkali plant. On Ball Lake, where mercury in perch averaged 0.36 mg/kg, no nests were initiated even though loons were present on territories. It should be noted, however, that nest initiation may have been affected by fluctuating water levels, given that nest initiation was not reduced at other lakes with higher concentrations of mercury in fish. This dietary concentration is very similar to the average concentration in six lakes on or adjacent to the Wabigoon – English River system, out of the flow of waterborne mercury but directly accessible to fish from the contaminated river system. Loons breeding in these six lakes (designated as C2 by Barr 1986) had lower productivity than loons nesting in control lakes, but higher productivity than loons nesting within the contaminated river system. Assuming a loon body weight of 4.0 kg and a food intake rate of 0.62 kg/day, a LOAEL of 0.056 mg/kg-day results for adult breeding loons based on Barr (1986). The average concentration of mercury in fish collected from 12 lakes adjacent to but independent of the contaminated river systems was 0.1 mg/kg; these lakes, collectively referred to as C4, were designated as the control population by Barr (1986). Indeed, there was no significant reproductive impairment reported for C4. Therefore, the 0.1 mg/kg dietary concentration was converted to a NOAEL of 0.016 mg/kg-day using the same body weight and food intake rate listed above. The TRV based on Barr (1986) is set equal to the geometric mean of the NOAEL and LOAEL—0.029 mg/kg-day. The test species dose of 0.029 mg/kg-day is generally consistent with the generic avian TRV of 0.026 mg/kg-day for methylmercury, reported in Nichols et al.'s (1999) review article, but is preferred over Nichols et al's (1999) value, due to its species-specificity. It is also more conservative than the LOAEL employed by Sample and Suter (1999) in their ERA of Clinch River/Poplar Creek watershed at Oak Ridge Reservation. The TRV generated from Barr's (1986) work is also more conservative than those generated from laboratory studies on other piscivorous bird species (e.g., Scheuhammer 1987, Frederick et al. 1997, Bouton et al. 1999). Chan et al. (2003) reported reproductive effects in birds at somewhat lower dietary concentrations (i.e., 0.1 mg/kg). Because several of the studies considered in Chan et al.'s (2003) review were conducted on nonpiscivorous bird species, however, the 0.1 mg/kg value is less applicable to this ERA than Barr's (1986) findings. Thus, the TRV of 0.029 mg/kg-day for effects of methylmercury on both loons and eagles is appropriate and sufficiently conservative for use in this ERA.

4.5.3 Mammalian TRVs for Methylmercury

Wobeser et al. (1976) conducted a chronic toxicity study on the effects of methylmercury on mink. Less applicable studies – such as those conducted on other mammals and those conducted using other forms of mercury – were excluded from consideration in this effects characterization. For example, two (Revis et al. 1989, Aulerich et al. 1974) of four mammalian studies cited by Sample et al. (1996) as potential sources upon which to base mercury TRVs were excluded from consideration based on these rationale.

Wobeser et al. (1976) fed adult female mink rations containing one of five dietary concentrations of methylmercury chloride over a 93-day period. Minor behavioural effects and histopathological abnormalities were observed in the lowest exposure group, fed 1.1 mg/kg methylmercury. Assuming a body weight of 0.69 kg and a food ingestion rate of 0.16 kg/day, this dietary concentration is equivalent to a LOAEL of 0.26 mg/kg-day. It is assumed that the NOAEL is ten-fold lower (i.e., 0.026 mg/kg-day), which yields a geometric mean of the NOAEL and LOAEL equal to 0.081 mg/kg-day. Given the study duration and species studied by Wobeser et al. (1976), no other UFs are warranted. Thus, the TRV selected to evaluate the effects of methylmercury on mammals is 0.081 mg/kg-day.

This TRV is supported by the broader literature on effects of mercury on mink, as well as regulatory decisions. For example, in a multi-generational study of female mink fed diets containing organic mercury-contaminated freshwater fish, Dansereau et al. (1999) reported that the dietary concentration of 1.0 mg/kg was the lowest observed effect level (LOEL) and that it was associated with decreased survival for first- and second-generation females. The Wobeser et al. (1976) study is also the basis for the wildlife value derived by USEPA (1995). USEPA (1997c) concluded that the appropriate LOAEL for effects of methylmercury on mink is 0.18 mg/kg-day. In their ERA for Clinch River/Poplar Creek watershed, Sample and Suter (1999) employed a slightly lower NOAEL and LOAEL (approximately 0.015 mg/kg-day and 0.12 mg/kg-day, respectively), but because the source(s) of those values are not provided, their appropriateness cannot be confirmed.

4.5.4 Avian TRVs for PCBs

As in the derivation of an avian TRV for methylmercury (Section 4.5.2), those studies that focused on the effects of PCBs on piscivorous species that reported ecologically pertinent endpoints (e.g., breeding and parenting behaviour, productivity) and yielded NOAELs and/or LOAELs were preferentially considered for the derivation of avian TRVs for PCBs. Tori and Peterle (1983) paired mourning doves that were fed 0 mg/kg, 10 mg/kg, or 40 mg/kg of Aroclor 1254 for 42 days and observed the courting and nesting behaviour of the doves for the next 30 days. Behaviours (e.g., perch coos, nest site selection, incubation) were scored and compiled for each courting and nesting phase. Doves fed 10 mg/kg PCBs spent a significantly increased number of days in the courtship phase ($p<0.01$), with only four of the eight pairs progressing into the nesting phase ($p<0.05$). These four nesting pairs took approximately twice as long to initiate nest building ($p<0.05$), which subsequently delayed egg laying. Using a food ingestion rate of 0.23 kg dry weight food/kg body weight-day and a body weight of 0.115 kg (USEPA 1993), a LOAEL of 2.6 mg/kg-day is calculated. Assuming the NOAEL is ten-fold lower yields a NOAEL of 0.26 mg/kg-day and a geometric mean of the NOAEL and LOAEL of 0.82 mg/kg-day, which is used in this ERA as the PCB TRV for common loons.

The TRV for bald eagles was based on a more closely related species, the American kestrel. Fernie et al. (2001, 2003) examined the reproductive effects of in ovo exposure to a 1:1:1 mixture of Aroclor 1248:1254:1260 in American kestrels. Adult kestrels were fed PCB-spiked food at 7 mg/kg-day for 100 days until their eggs hatched. Second generation kestrels were paired with unexposed kestrels with reproductive experience. Twenty-five percent of the in ovo PCB-exposed females failed to lay any eggs. Clutch initiation was delayed, and clutch sizes and fledging success were reduced in both male and female PCB-exposed birds. Due to the overall effect on reproductive success of the in ovo PCB-exposed kestrels and their taxonomic relatedness to the bald eagle, the unbounded LOAEL of 7 mg/kg-day and assumed NOAEL of 0.7 mg/kg-day are used to calculate a geometric mean value of 2.2 mg/kg-day, which is used in this ERA for the PCB TRV for bald eagles. Although there is uncertainty associated with the estimated NOAEL, comparisons to other studies demonstrates its conservatism, as discussed below.

Other researchers have evaluated PCB toxicity in non-piscivorous domestic and wild bird species. Such studies are discussed below to allow comparison with the selected TRVs, although they were judged less applicable than the work of Tori and Peterle (1983) and Fernie et al. (2001, 2003). Dahlgren et al. (1972) evaluated hatchability of eggs laid by adult ring-necked pheasants exposed to Aroclor 1254 for 16 weeks, at doses of 1.8 and 7.1 mg/kg-day. The higher dose caused reduced production and survival of offspring. At the lower PCB dose, a slight but statistically significant reduction in egg hatchability was noted during one of two trials. However, no significant effects on egg production or chick survival were observed, and the overall number of surviving chicks per hen was actually slightly higher than in the control group. Based on the overall effects on reproductive success, the LOAEL is identified as 7.1 mg/kg-day and the geometric mean of the NOAEL and LOAEL is 3.6 mg/kg-day, a result somewhat less conservative than Fernie et al.’s (2001, 2003).

In a study of Aroclor 1254 toxicity, Custer and Heinz (1980) fed mallards a diet containing 25 mg/kg for a month during breeding. No adverse effects were observed on the number of hens laying, date of the first egg laid, clutch size, fertility, hatching success, survival of ducklings to three weeks of age, or nest attentiveness. This study provides a NOAEL of 8.1 mg/kg-day, which is consistent with but slightly less conservative than Fernie et al.’s (2001, 2003) LOAEL. In a field study, Henning et al. (2003) found no adverse effects on reproduction of robins exposed to approximately 7.8 mg/kg-day total PCBs. This NOAEL is also consistent with but slightly less conservative than Fernie et al.’s (2001, 2003). McLane and Hughes (1980) monitored clutch sizes and hatchability in captive screech owls fed 3.0 mg/kg Aroclor 1248. No adverse effects were observed ($p>0.05$) at this dietary level, which corresponds to a dose of 0.41 mg/kg-day. Because this study only provides an unbounded NOAEL (i.e., no effects were observed at the highest concentration tested), it provides a less appropriate basis for a TRV than the studies by Tori and Peterle (1983) and Fernie et al. (2001, 2003).

4.5.5 Mammalian TRVs and Effect Concentrations for PCBs

Fuchsman et al. (2008) compiled information from 16 published studies evaluating effects of PCBs on mink reproductive success. More than 50 tests were included in the data set. Effects were assessed based on the number of surviving kits per mated female. Table 4-3 summarizes the available literature of the effects of PCB exposures on reproductive success of mink. Figure 4-1 illustrates the dose-response relationship for mink exposed to PCBs. Toxicity was defined as a reduction in productivity of more than 30%, as smaller effects were not reliably detectable based on experimental variability. Kit growth was also evaluated and was found to be a less sensitive endpoint. Exposures were expressed using various metrics, to compare which method best explained the variation in observed effects. The exposure metrics included measures of dietary and internal dose, as well as measures based on total PCBs and PCB congeners.

A TRV for this assessment is identified based on exposures measured as daily dietary intake of total PCBs. The lowest “toxic” dose in the data set (associated with approximately 50% reduction in reproductive success) is 0.057 mg/kg-day (Halbrook et al. 1999). However, this result does not appear to represent PCB-related toxicity, as 0.057 mg/kg-day was the lowest dose administered, and three higher doses resulted in no adverse effects (Halbrook et al. 1999). The highest NOAEL below this level is 0.053 mg/kg-day, from a study in which mink were fed PCB-contaminated seal blubber (Brunström et al. 2001). For comparison, a lack of reproductive toxicity was observed in other tests at a dose as high as 0.83 mg/kg-day (Käkelä et al. 2002), while the central tendency effect concentration to 50% of population tested (EC50) is estimated as 0.17 mg/kg-day (Fuchsman et al. 2008). Thus, there is a high degree of variability in the dose-response relationship when the dose is expressed in terms of total PCBs in the diet.

Although many studies have tested the effects of PCBs on mink reproduction, few studies have identified both NOAEL and LOAEL values. Approaches to identify a TRV from a single critical study yield results that are not consistent with a reasonably conservative interpretation of the overall dose-response relationship evident in Figure 4-1. Therefore, a TRV is identified for this ERA as 0.053 mg/kg-day, the highest dose at or below which toxicity has never been observed.

The level of uncertainty in the assessment of mink exposed to PCBs can be significantly reduced by accounting for the bioaccumulation potential in mink of site-specific PCB mixtures. This can be accomplished either on a total PCB basis (using PCB homologue concentrations in fish) or on a congener basis; both approaches are similarly successful in explaining observed variability in the dose-response relationship (Fuchsman et al. 2008). Both approaches are used as supplemental lines of evidence in this ERA. Analogous to the TRV identified above, effect concentrations to interpret estimated body burdens in mink are identified as the highest levels at or below which toxicity has never been observed. The effect concentration for total PCBs in mink is identified as 0.60 mg/kg, and the effect concentration for TEQ in mink is identified as 7.9 ng/kg.

4.5.6 Tissue Residue Guidelines

An additional line of evidence considered for piscivorous birds and mammals is the comparison of the distribution of fish tissue concentrations (both across species and by species) observed in the AOC to the TRGs for methylmercury and PCBs. These TRGs are intended to be protective of all piscivorous wildlife.

Environment Canada's (2002) TRG of 0.033 mg/kg for methylmercury was derived using the reference concentration (RC) of the Wilson's storm petrel, which is the most susceptible wildlife species evaluated by Environment Canada (2002) (it consumes almost its entire body weight in aquatic biota per day). Because the Wilson's storm petrel is a pelagic bird, it is not expected to forage in Lake Superior. Derivation of TRGs applicable to loons, bald eagle, and mink involves dividing the tolerable daily intake (TDI) for birds or mammals by food ingestion-to-body weight ratio (FI:bw ratio) for those species. Thus, using TDI and FI:bw ratio data listed in Environment Canada (2002), the TRG applicable to common loons is 0.031 mg/kg-day ÷ 0.18 = 0.17 mg/kg. The TRG applicable to bald eagles is 0.031 mg/kg-day ÷ 0.11 = 0.28 mg/kg. The TRG applicable to mink is 0.022 mg/kg-day ÷ 0.24 = 0.092 mg/kg. The lowest of these three TRGs, 0.092 mg/kg, is applied as the methylmercury TRG in this ERA in order to be protective of all three receptors.

Environment Canada's (1998) TRG for TEQs is equal to 0.79 ng/kg (Environment Canada 1998). It was derived from Wren et al.'s (1987) study on the effects of Aroclor 1254 on reproductive success in minks. Thus, interspecies conversion of the TRG for PCBs is not necessary, as it was for the TRG for methylmercury. Those fish tissue samples that were analyzed for PCB congeners were directly compared to the TEQ TRG. For all other fish tissue samples a total PCB-equivalent TRG was calculated as follows. Samples collected from nearby regions of Lake Superior as part of the Sportfish Contaminant Monitoring Program and the samples collected from Peninsula Harbour by EC and MOE in 2007 were analyzed for both congeners and total PCBs. Fish from Zone 8a (filename: Lake Superior – Peninsula H.xls, provided to ENVIRON by Environment Canada) and the 2007 fish data were used to develop a data set of 25 paired TEQ and total PCB results, to allow characterization of the relationship between TEQ and total PCB concentrations in local fish. The total PCB concentration was found to equal the TEQ concentration divided by 6.144×10^{-6}. Thus, a total PCB concentration of 0.13 mg/kg is extrapolated from the TRG for TEQs of 0.79 ng/kg (0.13 mg/kg = 7.9×10^{-7} mg/kg ÷ 6.114×10^{-6}).

5 ECOLOGICAL RISK CHARACTERIZATION

The fourth step of the ERA, risk characterization, integrates information derived from the three preceding elements in order to determine the potential for adverse effects in VECs as a result of exposure to mercury and PCBs in the AOC. In addition, the risk characterization describes the uncertainty associated with the risk estimates. As previously noted, Barnthouse et al. (2008) observes that "[r]egulations, policies, directives, and guidance documents frequently discuss the need for [ERAs] to consider risks to populations, not simply to individual organisms or organism-level attributes. The reason for this is that, from a management perspective, the population-level attributes such as abundance, persistence, age composition, and genetic diversity are usually more relevant than are the health or persistence of individual organisms" (Barnthouse et al.2008). Thus, if an ERA predicts unacceptable risks in ecological populations and communities, management actions are typically evaluated and often taken to mitigate such risks. Most often, however, ERAs evaluate only individual-level effects, due to the technical challenges and cost of conducting population-level assessments. If an ERA predicts unacceptable risks in individual organisms, management decisions generally consider the proportion of individual organisms at risk, the spatial scale of the impact, whether species at risk are protected, and whether acute effects are predicted. In contrast with ERAs, if unacceptable risks are predicted for individual human receptors in an HHRA, then management actions are typically evaluated and often taken to mitigate such risks because society places a high value on individual human lives.

As the foregoing sections describe, many of the lines of evidence employed in the ERA rely on comparisons of estimated exposure to effects levels (HQs). Such comparisons yield quotients, wherein values greater than one indicate that estimated exposures exceed effects levels and values less than one indicate that estimated exposures are below effects levels. Although HQs are not probabilities, higher HQs generally imply greater impacts to receptors. Within a given line of evidence, for example, an HQ of 0.5 would indicate acceptable ecological risks, while an HQ of 20 would indicate greater impacts than an HQ of 2. When interpreting HQs of, say, 2 vs. 20, it is critical that one considers the degree of conservatism used to estimate exposures and to derive effects levels. If precise, site-specific measurements and studies are used, then an HQ greater than one would be of greater concern than if modeled values or maximum measured exposures are compared to a screening effects level or conservative default effects level. By the same token, greater ecological risk is implied by a large proportion of exposure measurements exceeding an effects level with a strong scientific basis, as compared to relatively few exceedances of screening-level or default effects levels. For these reasons, it is critical that HQs greater than one be evaluated in the full context of the measurement endpoint uncertainty and relative to the scientific defensibility of all lines of evidence. Findings of different lines of evidence may contradict one another because different degrees of conservatism and uncertainty are inherent in different lines of evidence. Consequently, evaluation of the weight of evidence and uncertainty is a critical component of the overall ecological risk characterization.

It bears emphasizing that the benchmark of acceptable hazard – one – is typically expressed with one significant figure (i.e., the benchmark of acceptable hazard is 1, not 1.0 or 1.00), given the precision implicit in the many assumptions that are integrated in its calculation. In other words, HQs ranging from 0.0001 to 1.444 do not exceed the benchmark of acceptable hazard (1). While more than one significant figure may be displayed in HQ results for purposes of comparison, values between 1.0 and 1.4 are nonetheless interpreted as consistent with the benchmark of acceptable hazard.

5.1 Benthic Invertebrates

Section 4.3 describes the site-specific relationship between sediment chemistry, toxicity, and benthic community composition, based on an extensive investigation by Milani et al. (2002). The outcome of measurement endpoints identified for benthic invertebrates and mercury and PCB exposures are integrated in this section.

Site-specific benthic community structure. Benthic invertebrate community composition throughout JC differs from that of reference sites. The observed differences showed no relationship to mercury concentrations and were instead attributed to organic enrichment and water depth, rather than chemical contamination. Further analysis to distinguish the relative importance of organic enrichment versus water depth has not been conducted and most likely would not influence sediment management decisions. To the extent that the benthic community is influenced by organic enrichment, any sediment management actions would likely diminish such effects by covering or removing the organic material in the sediment.

Site-specific sediment toxicity. In subchronic and chronic laboratory tests with multiple species, very few sediment samples from the AOC exhibited any indication of toxicity. In most cases where toxicity was indicated, the observed effects were attributable to physical characteristics of the sediment. No relationship was observed between sediment toxicity and mercury concentrations, which were as high as 19.5 mg/kg in non-toxic sediment samples. PCBs were not measured in the toxicity study. The observed lack of toxicity confirms that differences in benthic community composition between JC and reference sites are due to factors such as water depth and organic enrichment, rather than mercury contamination. Considering that more than 90% of the surface sediment samples collected from JC contained mercury concentrations below 19.5 mg/kg, it is reasonable to conclude that sediment toxicity due to mercury is unlikely throughout JC and the rest of the AOC.

Comparison of sediment concentrations to guidelines and other data. As shown in Table 5-1, concentrations of mercury and PCBs exceeded low-end screening values in the majority of sediment samples collected from JC and RPH. The high-end screening values for total PCBs were infrequently exceeded and Ontario's high-end screening value (the SEL) for mercury was not exceeded in any sediment samples from RPH. As described in Section 4.3.3, these screening values do not represent cause-effect relationships and are most useful for initial screening purposes. Therefore, additional information on sediment toxicity from the scientific literature is also considered. The observed mercury concentrations in JC are similar to levels that have been found to be nontoxic at some other mercury-contaminated sites (Sferra et al. 1999), and the observed total PCB concentrations are well below levels at which PCBs are likely to be a primary cause of toxicity to benthic invertebrates (Fuchsman et al. 2006). Thus, the observed lack of toxicity of AOC sediment to benthic invertebrates is consistent with available information regarding the toxic potential of both PCBs and mercury in sediment.

Taken together, these lines of evidence indicate that benthic invertebrates in the AOC are not at significant risk due to mercury or PCBs in sediment. Because it is based on an extensive, site-specific biological investigation following BEAST, the ERA for benthic invertebrates entails little uncertainty. The likelihood of toxicity is not known for sediment samples containing higher mercury levels than those tested by Milani et al. (2002), but such samples represent only a small proportion of JC sediment (less than 10%). Similarly, the cause of observed toxicity is not known for a small number of sediment samples collected outside JC, but these samples again represent a minimal area. Overall, these uncertainties do not affect the conclusions of the ERA for benthic invertebrates.

5.2 Fish

Lines of evidence included in the risk characterization for fish are integrated below. Important uncertainties in this assessment are identified and discussed.

C*omparison of tissue concentrations to TRVs*. Table 5-2 summarizes the estimated risks posed to fish collected from the Peninsula Harbour AOC. Although results are presented separately for the overall AOC, JC and RPH, it should be noted that fish caught from JC and RPH likely represent the same population. As illustrated by the mean length of fish collected from JC and RPH (Table 5-2), it is likely that younger fish inhabit the shallower JC and move into RPH as they age. As previously noted, there are no barriers that would prevent fish from swimming between JC and RPH. Thus, findings of HQs greater than 1 in RPH but not in JC, as discussed below, likely reflect the movement of fish into RPH as they age and/or sampling bias (i.e., larger fish were sampled from RPH than JC), rather than differences in sediment concentrations in the two areas.

As shown in Table 5-2, the mean HQ for longnose sucker collected from RPH exceeds the target HQ of 1 (HQ=2). Because the mean HQ is a central tendency estimate, this finding suggests that adverse effects in longnose suckers from mercury may propagate to population-level effects. Although the mean HQs for mercury in no other species of fish exceed the target HQ of 1, the 95th percentile HQs for mercury exceed 1 for lake trout, walleye, and lake whitefish (as well as longnose sucker). Because the 95th percentile HQ is based on the most highly exposed individual fish, risks posed by mercury are predicted to be limited to individuals, rather than to propagate to population level effects in sportfish species. With the exception of longnose sucker, neither mean nor 95th percentile HQs for PCBs exceed 1 for any fish species. The 95th percentile HQs for longnose suckers exposed to PCBs are 2 in both the entire AOC and RPH, but is 0.4 in JC. Therefore, PCBs are predicted to pose unacceptable risks to individual longnose suckers, but are not other fish species at either the individual or population levels.

Fish community characterization. Recent data characterizing fisheries resources in JC were not designed to identify the occurrence or lack of toxicity to fish due to mercury and PCBs. As described in Section 4.4.1, fish in JC were less abundant but more diverse than in Carden Cove. No longnose suckers were sampled for chemical analysis from either JC or Carden Cove, limiting the ability to draw comparisons between the biological and chemical characterization of fish potentially at risk. The abundance of pollution-intolerant mottled sculpin and the occurrence of adult, juvenile, and young-of-year fish in JC are positive indicators of fish community health. However, the fisheries survey results cannot be considered conclusive with regard to subtle effects on the most highly exposed species. Therefore, this line of evidence receives less weight than the evaluation of tissue residues in the hazard assessment for fish.

Comparison of mercury and PCB tissue concentrations to local background concentrations. Fish tissue concentrations were compared to local background concentrations, as determined based on data from the SFCMP Data for Lake Superior were provided to ENVIRON by Environment Canada. Summary statistics were generated for samples collected from Zone 8a (Peninsula Harbour) and Zone 7, the monitoring block geographically closest to Peninsula Harbour. The results of this analysis are shown in Table 5-3. For longnose sucker, lake trout, and lake whitefish combined, as well as for longnose sucker alone, average fillet and extrapolated whole body concentrations of mercury and PCBs are higher in Peninsula Harbour than in Zone 7 (i.e. background). This trend is also apparent when median fillet concentrations are compared and when upper percentile fillet concentrations are compared. This comparison suggests that both regional sources (e.g., atmospheric deposition, other point sources) and localized sources (e.g., contaminated sediment in the AOC) contribute to bioaccumulation of PCBs and mercury in fish from the AOC.

Due to the limitations of the fisheries survey conducted for JC, comparisons to TRVs serve as the primary line of evidence for assessing risks to fish. Comparisons to local background concentrations do not, in themselves, shed light on the likelihood of adverse effects, but they do provide useful information on the contribution of regional versus site-specific sources of contamination. The integration of the available lines of evidence for assessing risks to fish is described below.

In summary, comparisons of tissue concentrations to TRVs indicate that reproduction may be impaired in sportfish (e.g., lake trout, walleye, lake whitefish) and bottom dwelling fish (e.g., longnose sucker) species in the AOC due to mercury and, in the case of longnose sucker only, PCBs. Risks are greatest in longnose sucker, where reproductive impairment may propagate to population level impacts. Although the potential for population level impacts in sportfish is less clear, risks are unacceptable for the most highly exposed individual lake trout, walleye, and lake whitefish. With the exception of longnose suckers, fish are not predicted to be adversely affected by PCBs. Available biological data are insufficient to confirm or refute conclusions based on comparisons of tissue concentrations to TRVs. Mercury and PCBs in AOC fish are likely attributable to a combination of regional and local sources. Several sources of uncertainty are identified in the risk evaluation for fish, as follows:

- *Fish tissue characterization in JC.* Concentrations of mercury and PCBs in fish tissue are better characterized for RPH than for JC. Walleye were not collected from JC and very few spottail shiners were collected from JC. Pink salmon collected from both JC and RPH were represented by small individuals (likely young-of-year). Larger pink salmon may contain higher concentrations of mercury and PCBs, since both are bioaccumulative. Lake trout are the most mobile of the fish species analyzed and individuals captured in JC may have been exposed to mercury and PCBs elsewhere. The locations from which fish samples were collected are not necessarily reflective of their primary foraging area. The observation that mercury concentrations in longnose suckers collected from RPH are higher than in those collected from JC is likely a function of smaller fish spending more time in JC and larger fish spending more time in RPH and/or sampling bias. The longnose sucker collected from JC averaged 28 cm in length, while those collected from RPH averaged 44 cm in length. Because mercury is bioaccumulative, larger fish tend to have higher tissue concentrations than small fish.

- *Fillet to whole body conversions.* Because most of the fish tissue samples from the AOC were analyzed as fillets, it was necessary to estimate whole body concentrations based on fillet data, as detailed in Appendix C. This conversion introduces relatively little uncertainty for mercury, but is less certain for PCBs. The conversion methods used for this assessment are intended to represent central tendency relationships between fillet and whole body concentrations; actual whole body concentrations may be somewhat higher or lower.
- *Toxicity reference values.* For mercury, the TRV is based on a tissue concentration that did not adversely affect reproduction or survival. For PCBs, the TRV is based on a test with the most toxic Aroclor formulation, in the absence of data identifying the site-specific composition of PCB mixtures in the AOC. Site-specific and species-specific toxicity thresholds may be somewhat higher or lower than the TRVs identified for this assessment.
- *Biological survey design.* In contrast to the benthic invertebrate study conducted for JC, the fisheries resource survey conducted by BEAK (2000, 2001) was not designed to determine whether adverse effects are occurring due to mercury or PCBs. The study is sufficient to determine that severe effects on fish reproduction and survival are not detectable, but it is inconclusive with regard to more subtle effects.

5.3 Risks to Piscivorous Wildlife

The primary measurement endpoints considered in the evaluation of risks to piscivorous wildlife were: 1) comparison of modeled dietary intake of mercury and PCBs by two representative avian species (common loons and bald eagles) and one representative mammalian species (mink) to doses reported in the literature as thresholds for adverse effects on survival or reproduction (i.e., HQs); and 2) comparison of whole body fish and invertebrate tissue concentrations of mercury and PCBs to TRGs. For the assessment of PCB-related risks to mink, estimated body burdens in mink were also compared to published effect concentrations as a supplemental line of evidence.

Although HQs are not probabilities, higher HQs generally imply more significant impacts to receptors. In evaluating the implications of HQs greater than one, the degree of conservatism used to estimate exposures and to derive effects levels is of paramount importance. HQs greater than one are evaluated in the full context of the measurement endpoint uncertainty and relative to the scientific defensibility of all lines of evidence. The benchmark of acceptable hazard – one – is typically expressed with one significant figure, in light of the known the precision of the inputs that are used in its calculation.

Table 5-4 presents HQs for common loon, bald eagles, and mink, based on both mean and 95% UCLM dietary concentrations and based on five scenarios reflecting varying proportions of the birds' diet from JC and RPH. The HQs reflect all available sampling data collected since 2000, including fish samples collected under the SFCMP for Zone 8a (Table 5-3). All calculated HQs for common loons are less than 1, regardless of whether the mean or 95% UCLM concentration is applied as the dietary concentration and regardless of the relative proportions of prey derived from JC and RPH. HQs for exposure of bald eagles to PCBs and mink exposed to mercury are also consistently less than 1, regardless of the dietary concentration applied and the relative proportion of prey derived from JC and RPH. HQs for exposure of bald eagles to mercury based on the mean concentration exceed 1 only when 100% of prey is derived from JC. However, when the 95% UCLM is applied as the dietary concentration, HQs exceed 1 for bald eagles exposed to mercury when 100% of prey is derived from JC and when 75% of prey is derived from JC and 25% from RPH. Under the most realistic yet appropriately conservative scenario (Scenario 4), the HQ for bald eagles exposed to mercury is less than 1. Thus, mercury is not predicted to pose an unacceptable risk to bald eagles and other piscivorous raptors under most likely conditions (as represented by Scenario 4). PCBs also are not predicted to pose an unacceptable risk to bald eagles and other piscivorous raptors.

HQs for exposure of mink to PCBs range from 1 to 9, depending upon the dietary concentration applied and the relative proportion of prey derived from each part of the AOC. When 25% or more of the prey is derived from RPH, the HQs are greater than 1. Given that habitat constraints along the shore of JC, mink are predicted to derive at least 25% of their prey from RPH. Thus, based on dose-based HQs, PCBs (but not mercury) pose an unacceptable risk to mink and other piscivorous mammals.

Table 5-5 presents body burden-based HQs for PCBs in mink. The HQs based on total PCBs and TEQs in mink agree within a factor of 5 and bracket the dose-based HQs. Estimated total PCB concentrations in mink yield higher HQs than estimated TEQs in mink. This difference may reflect the potential for toxicity due to PCB congeners other than the "dioxin-like" congeners for which TEQs are calculated. Alternatively, it may reflect analytical and model uncertainty. In any case, both total PCB and TEQ body burden estimates indicate that the site-specific composition of PCBs corresponds to a relatively high bioaccumulation potential, supporting the conservative TRV selected to interpret estimated dietary intake of PCBs by mink in this ERA. Because both dose-based HQs and body burden-based HQs for mink exposed to PCBs exceed 1, there is relatively low uncertainty in the conclusion that PCBs in Peninsula Harbour fish pose unacceptable risks to individual mink.

In August 2007, the suitability of the Peninsula Harbour AOC to support mink and/or river otter was evaluated (Appendix D). Although the presence of both species along the shoreline of the AOC was confirmed, the AOC is habitat-limited for piscivorous mammal populations. In particular, there is relatively little vegetation within 1 m of shoreline and relative few trees and shrubs within 100 m of the shoreline. Therefore, the AOC probably supports relatively few local mink and river otter, which constitute a small fraction of the regional population. Indeed, MNR found that only a few mink (i.e., 4 in 2002-2003, 1 in 2003-2004, 2 in 2004-2005) and one river otter (1 each in 2002-2003 and 2003-2004) were trapped in three different areas within Peninsula Harbour between 2002 and 2005. Thus, while PCBs pose an unacceptable risk to individual mink, the number of mink inhabiting the AOC is likely low due to habitat limitations. Consequently, adverse effects on individual mink are not predicted to propagate to population level impacts.

Compared to the HQ measurement endpoint, the TRG measurement endpoint indicated greater risk to mink. The majority (i.e. >50%) of available biota tissue concentrations exceed TRGs, indicating unacceptable risks are posed to piscivorous wildlife by current concentrations of mercury and PCBs in their prey. This finding generally holds across most species evaluated and regardless of whether the mean or 95% UCL concentration in tissue is compared to the TRG.

Sources of uncertainty associated with both measurement endpoints are summarized below. In the majority of cases, conservative assumptions are employed to compensate for unavoidable uncertainty in the endpoints. Key sources of uncertainty in the HQ measurement endpoint include:

- *Choice of Representative Receptors.* Representative wildlife receptors were selected based on their potential for maximum exposure and toxicological sensitivity to the mercury and PCBs. Thus, the three receptors (common loons, bald eagles, and mink) serve as conservative surrogates for all other wildlife likely to inhabit the AOC. That is, less exposed or less sensitive species in the same feeding guild may not be adversely affected in all cases where the representative receptors are predicted to be adversely affected. For example, although mink are predicted to be at risk from PCBs, it is uncertain whether racoons, for example, are also at risk.
- *Dietary Concentrations.* Analytical variability in the chemical analysis of prey tissues creates some uncertainty in the dietary concentration used in the exposure model. The potential direction of this uncertainty is unknown and could result in either under- or overestimation of exposure and risk. Appendix C details the data handling practices, which included size-normalization of fish tissue data and extrapolation of whole body tissue concentrations from fillet results. These practices were conducted in order to reduce uncertainty in the dietary concentration that could result from a sampling program that targeted fish consumed by human anglers, rather than by wildlife. In addition, smaller whole body fish samples were collected in 2007 to help address this limitation in the available data. Both size-normalization and extrapolations of whole body concentrations were based on the available species-specific and site-specific data; as such, they are expected to reduce uncertainty in the overall analysis. Consideration of both mean and 95% UCLM concentrations as dietary concentration allowed consideration of the effect of the statistical metric on the overall conclusions of the measurement endpoint; findings were not affected by the metric applied.
- *Dietary Composition.* In calculating dietary concentrations, sample results for fish species were averaged together in the proportion that they were represented in the sample data set. This practice implies that wildlife catch different species of fish in the same

proportions that different species were sampled in the field sampling program. It is not known whether a purposeful sampling design was employed or whether this assumption is realistic for the wildlife species considered. Therefore, this assumption could result in either an under- or overestimation of dietary concentration.

- *Area Use Factor.* In order to account for varying proportions of the diet derived from JC and RPH, as well as the variable use of the two areas by fish, five wildlife scenarios were evaluated using different AUFs. Scenario 4 was judged most realistic, yet appropriately conservative scenario for common loon, bald eagle, and mink, given the much larger area of RPH as compared to JC. The conclusions for bald eagles exposed to mercury and the mink exposed to PCBs were affected by the AUF. Thus, in these cases, the foraging range may be a source of uncertainty.
- *Body Weight*. The free metabolic rate equations (used in the estimation of food ingestion rates) require body weight as an input variable. Because body weights of wildlife receptors actually inhabiting the AOC were not available, literature-derived values were employed. While body weights of bald eagles and mink do not vary a great deal across North America, those of loons are highly variable. Because body weight is one of the variables used to estimate the food ingestion rate, and because exposure is closely tied to food ingestion rate, HQs for common loons may be either under- or overestimated if the loons inhabiting the AOC weigh substantially more or less than the 4.0 kg assumed in this ERA.
- *Toxicity Reference Values*. The greatest source of uncertainty in the assessment of risk to bald eagles from both mercury and PCBs and in risks to loons from PCBs is the lack of toxicity studies involving this species. Considerable interspecies variability in sensitivity to mercury and PCBs is evident from the review of the ecotoxicological literature. For example, grainivorous birds (such as chickens and pheasants) appear to be considerably more sensitive to the toxicological effects of PCBs than are piscivorous birds. In order to address this uncertainty, critical studies were selected from the most closely related species for which a high quality study had been conducted. Because the mercury TRV applied to loons was based on a study conducted on loons and because both the mercury and PCB TRVs applied to mink were based on studies conducted on mink, interspecies variability in sensitivity to these chemicals is unlikely to affect certainty in these HQs. Also because TRVs were based on the geometric mean of the NOAEL and LOAEL, the selected TRVs may represent a dose that is less than the true threshold effect level. Consequently, HQs equal to 1 may not reflect likely adverse effects.
- *Population Level Effects*. The endpoints evaluated focus on effects on individual organisms, rather than populations. Through natural compensatory mechanisms, many populations are generally sustainable, even if some individuals (e.g., less than 20%) are affected. Thus, predicted effects at the individual organism level do not necessarily translate into adverse, ecologically relevant, impacts at the population level.

Key sources of uncertainty in the TRG measurement endpoint include:

- *Dietary Concentrations.* The assumed dietary concentrations were unavoidably influenced by the field sampling programs that generated the underlying data. To the extent that certain species or size classes were either under- or overrepresented relative to wildlife's prey preferences, some uncertainty may be introduced into the resultant HQs. However, data are relatively sparse on the species preferences of prey for these wildlife, so it is not feasible to weight the calculated dietary concentrations by species preferences. While it is not possible to quantify the amount of uncertainty introduced by the dietary concentrations, it is unlikely that improved accuracy in the dietary concentrations would substantially affect the overall conclusions of the ERA, given that findings were generally consistent when either the mean or 95% UCLM was used as the dietary concentration. The data handling practices discussed above (i.e., size- normalization and extrapolation of whole body concentrations from fillet data) were applied to the biota tissue concentrations that were compared to TRGs. Again, these practices were conducted in order to reduce uncertainty in tissue concentration that could result from a sampling program that targeted fish consumed by human anglers, rather than wildlife prey. Consideration of mean concentrations, 95% UCLM concentrations, and the percent of samples exceeding criteria allowed consideration of the effect of the statistical metric on the overall conclusions of the measurement endpoint; findings were not affected by the metric applied.

- *Tissue Residue Guideline.* The TRGs applied in this analysis were developed by Environment Canada (1998, 2002) and are detailed in scientific supporting documents. Through the use of safety factors and the choice of critical studies, the TRGs incorporate greater conservatism than do the TRVs developed in this ERA for use in the HQs.

In conclusion, uncertainty is associated with both measurement endpoints, although greater conservatism is built into the TRGs than the TRVs used for the HQ calculations. There are multiple additional sources of conservatism in the HQs. Thus, while the TRG measurement endpoint is more conservative than the HQ measurement endpoint, it does not appear that the HQs underestimate risks. Thus, the only wildlife receptor potentially at risk based on this analysis is the mink, due to exposure to PCBs. For this receptor, population-level effects are not predicted due to habitat limitations along Peninsula Harbour shoreline.

6 SCREENING LEVEL HUMAN HEALTH RISK ASSESSMENT

The objective of this screening level HHRA is to quantitatively evaluate the potential for risks to human health from the consumption of fish caught in the AOC, based on currently available data. While potential risks are evaluated in this section in a streamlined manner, general methodologies and specific assumptions are consistent with HHRA practices as described by MOEE (1996a,b) and MOE (2005a).

MOEE (1996a) defines HHRA as "the evaluation of the probability (including likelihood and severity) of adverse health consequences, and the accompanying uncertainties, to humans caused by the presence of a chemical at a given site…The first step involves formulating the problem based on the nature and extent of contamination in the media and locations of concern. Routes of contaminant transport due to site specific characteristics must be accounted for. The site is characterized to determine what receptor populations are currently present at or near the site…Hazard identification results in a preliminary identification of potential human receptors, potential exposure pathways and potential impact on human health…Hazard identification/problem formulation is followed by the determination of the risk associated with the presence of the chemical to the receptors identified at or near the site. This risk is estimated in three major steps: toxicity assessment, exposure assessment and risk characterization."

The following subsections describe the exposure settings and key exposure scenarios, methods for estimating human doses of methylmercury and PCBs as a result of consuming fish from the AOC, toxicity values for methylmercury and PCBs, risk estimates, and key sources of uncertainty.

6.1 Exposure Assessment

Exposure assessment is the process of measuring or estimating the intensity, frequency, and duration of human exposure to substances present in the environment. The objectives of the exposure assessment are to estimate exposure point concentrations (EPCs) and doses for each relevant exposure scenario. This screening level HHRA focuses on the fish consumption pathway. Other potential exposure pathways, such as dermal contact or incidental ingestion of contaminated sediment, are outside of the scope of this evaluation. For bioaccumulative chemicals in aquatic systems, the fish consumption pathway is generally a far greater contributor to overall exposure than all other pathways combined. Key guidance considered in quantifying exposure to humans include MOE (2005a), USEPA (1989a, 1989b, 1991, 1992b, 1997b, 2000), HC (1994, 2004a, 2004b), and MOEE (1996a,b).

The overall CSM for human exposures to methylmercury and PCBs in AOC fish is illustrated in Figure 6-1. In this screening level screening level HHRA, human exposures to methylmercury in fish tissue are quantified for sport anglers, using the following four age subgroups as recommended by HC (2004a): toddler (7 months through 4 years), child (5 through 11 years), adolescent (12 through 19 years), and adult (greater or equal to 20 years). Pregnant women are another population subgroup that is highly sensitive to exposure to methylmercury and PCBs in fish. Toxicological studies recommend the use of a similar TDI[4] for pregnant women and children (HC 2004c). Because the evaluation of sensitive pre-adult subgroups is expected to be protective of pregnant women, pregnant women are not quantitatively evaluated in this screening level HHRA. Although infants are not expected to consume any fish directly, they may be exposed to methylmercury and PCBs via breast milk. Infant exposures also are not quantitatively evaluated in this screening level HHRA. Rather, it is assumed that lactational exposures to infants are comparable to or greater than exposures to toddlers via fish consumption.

Given the available data on concentrations of methylmercury and PCBs in fish tissue, the exposure assessment is based on the central tendency or average exposures for sport anglers. In order to ensure that the EPC is not underestimated due to variability or uncertainty in the available analytical data for fish tissue concentrations, the lower of the 95% UCLM and the maximum concentrations of mercury and total PCBs in fish fillets are employed as EPCs in this analysis. Each of the potentially exposed populations is described in more detail below.

6.1.1 Site-Specific Fish Consumption Survey

In order to provide site-specific estimates of fish consumption rates from Peninsula Harbour, a fish consumption survey was conducted in the town of Marathon and the nearby Pic River Indian Reserve. The survey was carried out in early 2008 by EcoSuperior, with the support of Environment Canada and MOE. The purpose of the survey was to determine whether sport anglers or subsistence anglers are consuming fish from the AOC, and if so, which fish species are targeted and how much fish is consumed. A summary of survey results is provided below. Appendix F provides a detailed description of the survey methods and results.

A survey asking about fishing and fish consumption habits over the previous year was distributed by mail to every household in Marathon. Residents were asked to fill out the survey and return it using a postage-paid envelope. Of the 1,310 surveys mailed to Marathon residents, 221 (17%) were returned. On the Pic River Reserve, surveys were distributed through door-to-door canvassing. Of the 192 households in the Pic River Reserve, 20 (10%) returned a survey.

Most households (84%) reported eating sport-caught fish, and a significant number (17%) reported eating fish caught from the Peninsula Harbour AOC. Although fishing advisories for Peninsula Harbour have been published by MOE (2005b) and the MNR (2004), none of the survey respondents mentioned the fish advisory as a reason for limiting fishing in the AOC. The survey results suggest that the existing fish advisories are not widely followed. There was no evidence in the survey responses that there are subsistence anglers using the AOC.

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[4] As detailed in Section 6.3 below, the TDI is defined as an estimate of a daily exposure level for a human population that is likely to be without an appreciable risk of adverse effects during a lifetime. The TDI is a type of TRV used to characterize threshold contaminants.

Respondents with the highest reported fish consumption rates also typically reported fishing mostly outside of the AOC. Households reported eating a variety of sport-caught fish including salmon, lake trout, walleye, and whitefish.

6.1.2 Potentially Exposed Populations

Sport anglers include avid anglers and their household or family members, who are assumed to consume fish caught recreationally from the Peninsula Harbour. Based on the results of the fish consumption survey, sport anglers are the only potentially exposed population that warrants evaluation in this screening level HHRA. Sport anglers fishing from a boat or from rocky areas on shore are unlikely to be significantly exposed to methylmercury or PCBs via dermal contact or incidental ingestion of sediment. Thus, pathways other than fish consumption are not considered in this screening level HHRA.

Subsistence anglers are individuals who consume fish as a significant portion of their diet. Although the presence of subsistence anglers in the Great Lakes is supported by several studies (Richardson and Currie 1993, Peterson et al. 1994, Gerstenberger et al. 1997), there was no evidence from the fish consumption survey that subsistence anglers are targeting fish from Peninsula Harbour. Because First Nations people are among the ethnic groups that sometimes fish for subsistence, the closest First Nations community to the AOC, the Pic River First National Reserve #50, was included in the fish consumption survey conducted by EcoSuperior. The Pic River Reserve is located about 18 km southeast of Marathon and 3 km east of Lake Superior. Although the response rates from the Pic River Reserve were low, there were no responses indicating high levels of fish consumption from Peninsula Harbour. Thus, this screening level HHRA does not evaluate risks to subsistence anglers targeting fish in Peninsula Harbour.

6.2 Dose Estimation

Human doses of methylmercury and PCBs are estimated in a manner consistent with MOE (2005a) and HC (1994, 2004a) guidance. In cases where the MOE and HC guidance lack recommendations for specific assumptions, USEPA (1991, 1997b, 2000) guidance is also considered, factoring in local considerations and updates to relevant studies that have been released since publication of the guidance documents.

6.2.1 Dose Equation

As discussed above, this analysis focuses on intake of methylmercury and PCBs through consumption of locally caught fish. The equation employed is:

Eqn. 7

$$EDI = \frac{C_{fish} \times CL \times FCR \times FI \times EF \times ED}{BW \times AT} \times 10^{-3} \frac{kg}{g}$$

Where:

EDI	=	estimated daily intake (mg/kg-day)
C_{fish}	=	exposure point concentration in fish tissue (mg/kg)
CL	=	cooking loss factor (unitless)
FCR	=	fish consumption rate (g/day)
FI	=	fraction ingested from contaminated source (unitless)
EF	=	exposure frequency (day per year or day/year)
ED	=	exposure duration (years)
BW	=	body weight (kg)
AT	=	averaging time (days)

6.2.2 Exposure Factor Values and Basis

The values selected for all exposure factors are discussed below and are summarized in Table 6-1.

6.2.2.1 Exposure Point Concentrations

USEPA (1989a) defines the EPC as the representative chemical concentration a receptor may contact over the exposure period. In order to estimate an average or central tendency dose, the EPC should be based on the mean fish tissue concentration. However, in order to ensure that the mean concentration is not underestimated due to variability or uncertainty in the available analytical data, the lower of 95% UCLM and the maximum concentrations of mercury and total PCBs in fish fillets are employed as EPCs in this analysis. Because humans typically consume fish that are at least 15 cm in length, only samples from fish this size or larger were considered in the calculation of the EPC. Exclusion of smaller fish from the EPC calculation is conservative, in that larger fish typically contain higher concentrations of methylmercury and PCBs than smaller fish. Because only limited fish samples collected from the AOC were analyzed for PCB congeners (which are necessary to calculate TEQ concentrations), this screening level HHRA evaluates PCB exposures based on total PCBs, but not TEQs. The effect of this practice is discussed in the uncertainty analysis. EPCs are listed in Table 6-2. EPCs were calculated using results from fish specifies targeted by sport anglers, as reported in the site-specific fish consumption survey. The target species with available sampling results are lake trout, walleye, and lake whitefish.

6.2.2.2 Cooking Loss Factor

The cooking loss factor accounts for the possible reduction in chemical concentrations in fish tissue as a result of cooking and food preparation. The cooking loss factor is defined as the fish tissue concentration after cooking divided by the fish tissue concentration before cooking. Separate cooking loss factors were estimated for methylmercury and PCBs.

For methylmercury, skinning, trimming, and cooking the fish does not significantly reduce the concentration in the fillet (USEPA 2001). Thus, a cooking loss factor equal to 1 is assumed for methylmercury.

As reported in USEPA (2000), the weighted average PCB reduction due to cooking of fish from the Great Lakes is equal to 30.3%, based on the Zabik et al. (1994) study. Thus, the cooking loss factor for PCBs is equal to 0.7 (one minus the cooking loss). At least 14 other studies (e.g., Reinert et al. 1972, Trotter et al. 1988, Zabik et al. 1979, 1995a,b, 1996) have quantified the loss of PCBs and other chemicals during cooking through the relatively simple study design of analyzing paired raw and cooked fish tissue samples and reporting the differences in concentrations. Wilson et al. (1998) critically reviewed 14 studies that employed this design for PCB analyses, and summarized findings for the highest quality studies (i.e., those listed above) by cooking method and by percentile. Averaging the median PCB losses for each cooking method reported in Wilson et al. (1998) results in a cooking loss equal to 40.6%, which generally agrees with the value recommended by USEPA (2000). The more conservative value (i.e., from USEPA 2000) is employed in this screening level HHRA.

6.2.2.3 Fish Consumption Rate

The fish consumption rate for the adult sport angler was estimated from the rates of adult respondents to the site-specific fish consumption survey who eat at least some sport fish. To provide a conservative estimate that is protective of most of the population, we selected the 80th percentile of the reported rates (17.5 g/day). This is generally consistent with the average rate of 21.3 g/day reported by a similar survey conducted in the rural town of Cornwall, Ontario (Kearney et al. 1995). HC (2004a) does not provide default fish consumption rates for sport anglers.

Fish consumption rates for pre-adult household members of sport anglers are adjusted by a ratio equal to the fish consumption rate of the age cohort of interest divided by the adult fish consumption rate, as reported for the general population by HC (2004a). The extrapolated fish consumption rates for toddler, child, and adolescent household members of sport anglers are equal to 8.8 g/day, 14.2 g/day, and 16.4 g/day, respectively.

6.2.2.4 Fraction of Fish Ingested from the Source

The fish consumption rates discussed above reflect all sport-caught freshwater fish consumed, including those caught from water bodies other than Peninsula Harbour. As such, they overestimate actual consumption from Peninsula Harbour. The fraction of the total sport-caught fish consumed that is from the AOC was estimated using the median value (0.2) reported in the fish consumption survey by anglers fishing at least some of the time in the AOC.

6.2.2.5 Exposure Frequency

Because the fish consumption rates (described above) are estimated as annual values in grams per day, exposure frequency is expressed on an annualized basis (i.e., 365 days per year) for all population categories.

6.2.2.6 Exposure Duration

Exposure duration describes the amount of time (in years) that an individual might fish Peninsula Harbour on a regular basis. The default exposure duration recommended by USEPA (1991) – 30 years – is assumed for the adult population categories. Exposure duration for other age cohorts is equal to the duration of the age range (i.e., 3.5 years, 6 years, and 7 years for the toddler, child, and adolescent cohorts, respectively).

6.2.2.7 Body Weight

Consistent with HC (2004a), this screening level HHRA employs body weights equal to 16.5 kg, 32.9 kg, 59.7 kg, and 70.7 kg for the toddler, child, adolescent, and adult age cohorts, respectively.

6.2.2.8 Averaging Time

The averaging time depends on whether non-threshold or threshold health effects are being evaluated. As discussed in Section 6.3, both PCBs and methylmercury are considered threshold contaminants, meaning that a biological threshold exists below which no health effects are expected (HC 2004a). For threshold contaminants, the averaging time for evaluating hazards is set equal to the exposure duration (HC 2004a).

6.2.3 Estimated Daily Intakes

Estimated daily intakes (EDIs) of methylmercury and PCBs associated with the consumption of fish caught in the AOC are calculated and presented in Table 6-3. EDIs for sport anglers are based on EPCs calculated from walleye, lake trout, and lake whitefish. Because the methylmercury and PCBs are considered threshold contaminants, EDIs are proportional to fish consumption per unit body weight and independent of averaging time. Thus, the age cohort with the highest fish consumption per body weight – toddlers – has the highest EDIs, followed by children, adolescents, and adults.

6.3 Toxicity Assessment

TRVs describe the relationship between the extent of exposure to chemicals and the likelihood and/or severity of adverse health effects. Types of TRVs can be divided between those used to quantify threshold effects and those used to quantify non-threshold effects. For non-threshold effects, it is assumed there is some probability of harm at any level of exposure. Consequently, it is not possible to determine a dose below which adverse effects do not occur. HC (1994) only considers mutagenesis and genotoxic carcinogenesis as non-threshold effects; other adverse health effects are considered to be threshold effects. TRVs for non-threshold effects are typically presented as slope factors, unit risk factors, or risk specific doses. In contrast, for substances causing threshold effects, a minimum dose can be defined below which the effect will not occur. The TDI is the TRV most often used to evaluate threshold effects resulting from exposures to the chemical in question. The TDI is defined as an estimate of a daily exposure level for a human population that is likely to be without an appreciable risk of adverse effects during a lifetime.

According to HC (1994) guidance, cancer effects are evaluated as either threshold effects or as non-threshold effects, depending on the strength of the evidence that the substance is a human carcinogen or germ cell mutagen. The carcinogenicity and germ cell mutagenicity of a chemical is classified into six categories (denoted Group I through Group VI) by HC (1994). Only chemicals that are classified as Group I (Carcinogenic to Humans/Human Germ Cell Mutagen) or Group II (Probably Carcinogenic to Humans/Probable Human Germ Cell Mutagen) are evaluated using the non-threshold approach described by Canadian Council of Ministers of the Environment (CCME) (2006). Since neither methylmercury nor PCBs has been classified as Group I or Group II carcinogens, TRVs based on a threshold model are used in this screening level HHRA.

Following the guidance of CCME (2006), TRVs published by HC were first considered for use in this screening level HHRA. If a TRV was not available from HC or for some reason was not appropriate, TRVs from other agencies, including USEPA and the World Health Organization (WHO) were considered. The TRVs used in this screening level HHRA are listed in Table 6-4. The following sections describe the basis for these values in more detail.

6.3.1 Mercury and Methylmercury

Mercury is transformed into methylmercury when the oxidized or mercuric species gains a methyl group. The methylation of mercury is primarily a natural, biological process that results in the production of methylmercury compounds that are both more toxic and more bioaccumulative than mercury salts or elemental mercury. In humans, methylmercury targets the central nervous system, as well as the cardiovascular, the immune, and the renal systems (HC 2004c). Since 90% to 100% of the mercury in most fish is present as methylmercury (USEPA 2001), this screening level HHRA assumes that the total mercury concentration measured in fish represents methylmercury. The screening level HHRA then evaluates human health risks from ingestion of methylmercury due to fish consumption. It is not necessary to evaluate the risk from total mercury separately because elemental mercury is much less toxic than methylmercury. Thus, the assumption that all the mercury consumed is methylmercury is protective of human health.

This screening level HHRA uses the methylmercury TDI of 0.00047 mg/kg-day for adults (HC 2004b) and the TDI of 0.0002 mg/kg-day for pre-adult age cohorts (HC 2004c). These values are similar to but slightly less conservative than the reference dose of 0.0001 mg/kg-day recommended by USEPA (USEPA 2006).

6.3.2 Polychlorinated Biphenyls

Adverse effects associated with PCBs include liver, thyroid, dermal and ocular changes, immunological alterations, neurodevelopmental changes, reduced birth weight, reproductive toxicity, and cancer (ATSDR 2000). Quantifying the toxicity of PCBs is complicated by the fact that PCBs are not a single compound, but rather constitute a class of compounds comprising 209 different congeners. Of these congeners, the most toxic are the 12 coplanar or "dioxin-like" congeners, which have a mode of action and structure similar to 2,3,7,8-TCDD. Most, if not all, toxic and biological effects of these dioxin-like PCBs are mediated through the same receptor and thus have a common mechanism of action (Van den Berg et al. 2006). Very few fish samples from the AOC were analyzed for dioxin-like PCBs. Thus, the health risk of total PCBs, rather than TEQs, is the focus of this screening level HHRA. As described below, health effects of total PCBs are evaluated using threshold TRVs shown in Table 6-5.

Standards Development Branch (SDB) of MOE has recently developed an interim TRV for total PCBs for use in site specific HHRAs (pers. comm. Dr. Paul Welsh, December 4, 2006). This interim TRV is based on adopting the TRV recommended by the WHO for PCBs, following a detailed review and analysis of the most up-to-date TRVs for PCBs proposed by recognized regulatory/health agencies. The interim TRV has undergone peer review and is in the process of being finalized. In the meantime, SDB recommends that for current risk assessments, PCBs are assessed as total PCBs using the TRVs recommended by the WHO (pers. comm., Dr. Paul Welsh, December 4, 2006). The WHO value of 2.0×10^{-5} mg/kg-day is the same as the reference dose recommended by the USEPA for evaluating noncancer effects from Aroclor 1254, the most toxic of the commercial PCB mixtures (USEPA 2006). Following the recommendations of MOE, this screening level HHRA employs this TDI to evaluate health effects from total PCBs. An analysis indicating that the mode of action responsible for the carcinogenicity of PCB mixtures is consistent with a non-genotoxic mechanism, and hence with the presence of a biological threshold is summarized in Balagopal et al. (2005). In contrast, USEPA recommends evaluating the cancer effects of total PCBs using a non-threshold approach (i.e., using a cancer slope factor).

6.4 Risk Characterization

Risk characterization is the final step in the HHRA process. In this step, the results of the exposure assessment and toxicity assessment are integrated to yield quantitative measures of hazard associated with the fish consumption pathway.

HQs are quantitative estimates of threshold health effects. The HQ is the ratio of the EDI dose and the appropriate threshold TRV (i.e., the TDI), as presented below for chemical is:

Eqn. 8

$$HQ_i = \frac{EDI_i}{TDI_i}$$

If the HQ exceeds one, then the exposed population is at risk for adverse health effects from ingestion of fish from the AOC. MOE (2005a) recommends that no more than 20% of the TDI should be apportioned to each environmental medium. Although EDIs associated with other media (e.g., groundwater, soil, air, surface water) have not been calculated for this site, based on experience at other sites with PCBs and/or methylmercury in fish tissue, risks associated with other media are expected to be minor compared to fish consumption. Indeed, given that fishing in Peninsula Harbour is most likely to occur from boats, docks, or rock outcrops along the shore, it is unlikely that anglers would be exposed to mercury or PCBs in sediment. Because both chemicals have a high affinity for organic matter, neither is likely to be present in surface water at detectable concentrations. Therefore, anglers' exposure to mercury and PCBs via incidental ingestion of sediment or surface water and dermal contact with sediment and surface water are expected to be negligible. For these reasons, a target HQ of 1, rather than 0.2, is applied in this screening level HHRA. An exceedance of the target HQ of 1 indicates potential hazard, but it does not reflect the probability of an adverse effect nor does it necessarily imply that adverse health effects will occur. Since methylmercury and PCBs do not share a common mechanism of action for most toxicity endpoints, HQs are not aggregated across chemicals.

Table 6-5 lists HQs associated with consumption of fish caught from the AOC.

6.4.1 Methylmercury

For methylmercury, HQs for sport anglers do not exceed the target value of 1 for any age cohort. The maximum HQ of 0.2 is associated with a toddler eating fish from JC. Since all of the methylmercury HQs are well below 1, no adverse health effects are expected from exposure to methylmercury via consumption of fish from the AOC by sport anglers.

6.4.2 Polychlorinated Biphenyls

For total PCBs, the sport angler HQs exceed the target of 1 for all scenarios, except for adolescents and adults consuming fish from either the entire AOC or JC only. The maximum PCB HQ is 5 for a toddler consuming fish from RPH. Based on this analysis, there may be a risk of adverse health effects to sport anglers that consume fish from the AOC, particularly the portion of the AOC outside of JC. However, if HQs are calculated using the mean fish concentrations (shown in Table 6-2) instead of 95% UCLMs, then all of the PCB HQs are 1 or less, indicating that risks are acceptable. Thus, there is uncertainty as to whether PCB concentrations in fish are above levels of concern.

By analogy to the toddler HQ, hazards to infants and pregnant women are also expected to exceed the target HQ of 1. As previously discussed, because infants are not assumed to eat any fish (HC 2004a), HQs are not calculated for this age cohort. However, hazards to infants resulting from lactational exposure (i.e., breastfeeding) may be comparable to or greater than those posed to toddlers from fish consumption. Similarly, hazards to pregnant women are assumed to be comparable to or greater than those posed to toddlers and children.

It is important to note that there are health benefits, as well as health risks, to infants as a result of maternal consumption of fish during pregnancy and while breastfeeding. A comprehensive review of the balance between the benefits and risks from fish consumption to populations of all ages was recently performed by the U.S. Institute of Medicine (2007). There are several observational studies that demonstrate the net benefit on child development from consumption by lactating woman of fish with typical levels of contamination (Daniels 2004, Oken 2005). In addition to benefits from fish consumption, there are also well established benefits to child cognitive development from breastfeeding itself, as opposed to bottle-feeding (Anderson et al. 1999). In observational studies of Faroe Island residents, who consume fish with highly elevated levels of methylmercury, the typical substantial benefits of breastfeeding were not observed (Jensen et al. 2005). The authors speculated that the adverse effects of the relatively high methylmercury exposure mitigated the benefits of breastfeeding. However, since there was no evidence of net adverse effects, the authors concluded that the Faroe Island residents could still safely breastfeed their infants. Thus, while the risk estimates for the Peninsula Harbour AOC suggest the potential for adverse health effects in infants from lactational exposures, there may still be net health benefits to infants in the AOC from breastfeeding and maternal fish consumption.

6.5 Sources of Uncertainty

Some variability and uncertainty are inherent in every HHRA because of the data and assumptions used in the assessment. The primary goal of uncertainty analysis is to describe the extent to which the hazards may be over- or underestimated, and to identify the specific uncertainties and conservatisms associated with the hazard estimates. Formal quantitative analyses of variability or uncertainty were not conducted in this screening level HHRA; however, major sources of variability and uncertainty are identified and considered to the extent that they would affect the conclusions drawn from this screening level HHRA.

There is uncertainty in the estimates of PCB and methylmercury concentrations in fish consumed by sport anglers. The sport angler risk evaluation was based on the 95% UCLM, a conservative estimate of the mean fish concentration. As discussed above, if the risk evaluation was based on the mean fish concentration instead of the 95% UCLM, then the estimated hazards would be lower and generally below levels of concern.

Due to the limited number of samples analyzed for congeners, health hazards posed by exposure to dioxin-like PCBs were not evaluated. Depending on the distribution of PCB congeners present in fish tissue caught from the AOC, the hazard from PCBs estimated in the HHRA may either over- or under-estimate the true hazard. However, since the PCB HQs are generally greater than the target HQ of 1, the evaluation of dioxin-like PCBs using the TEQ approach would not change the conclusions of the screening level HHRA.

The screening level HHRA relies on default values for exposure parameters, such as exposure duration and body weight. Modification of these assumptions would have a linear effect on the estimates of hazard. For example, increasing the body weight would result in a decrease of the HQ. The selected values are based on average values provided by HC (2004a). If body weights or other exposure parameters of local residents are significantly different than the values used in the assessment, health hazards could be under- or overestimated.

Another source of uncertainty in this screening level HHRA is the estimates of fish consumption rates and the fraction of fish ingested from source. Various guidance documents offer a broad range of fish consumption rates. Health hazards for particular receptors may be over- or underestimated based on lower or higher ingestion rates. The adult fish consumption rate and fraction ingested from source was based on a site-specific fish consumption survey. The survey required respondents to estimate the number of fish meals and average portion sizes for the previous year. Given the difficulty of accurately recalling these quantities, there is some uncertainty in the fish consumption rates derived from the survey results. In order to provide a conservative estimate, the 80th percentile of reported fish consumption rates was used in this assessment. Because health hazards are linearly related to fish consumption, different fish consumption rate assumptions will affect calculated HQs in an analogous manner.

Another source of uncertainty is the cooking loss factor. Different cooking methods (baking, boiling, broiling, frying, smoking, microwaving) can result in different percent reductions of PCB concentrations in fish that can range within a factor of three (Wilson et al. 1998). The cooking loss factor used in this screening level HHRA for PCBs is an average recommended by USEPA (2000) that is approximately two-fold lower than the average cooking loss reported for boiling and about 16% greater than the average cooking loss reported for baking (Wilson et al. 1998). This assumption may result in either over- or underestimation of HQs, depending on preferred cooking methods. However, given the very high HQs for PCBs estimated for all population subgroups, alternative assumptions regarding cooking methods and resultant cooking losses of PCBs would not change the conclusions of the screening level HHRA.

The TRVs for total PCBs are based on studies using a limited set of PCB mixtures – mostly Aroclors 1254 and 1242 – which are among the most potent to humans. Health hazards from exposure to PCB mixtures that have lower chlorine contents are likely to be lower. However, PCB mixtures with a higher content of more toxic congeners, notably highly chlorinated non- and mono-ortho congeners, may pose a larger health hazard. While PCB congener data are available for only one fish tissue sample, more than 50 sediment samples from the AOC have been analyzed for PCB congeners. The congener composition of PCBs in sediment generally indicates a higher percentage of penta-, hexa-, and heptachlorobiphenyls, as compared to the less-chlorinated tri- and tetrachlorobiphenyls. Although the composition of PCBs in fish tissue is likely to differ somewhat from that in sediment, the sediment data indicate an overall PCB mixture composition similar to Aroclor 1254. Thus, the comparability between exposure and effects data for PCBs is likely reasonable.

The target HQ of 1 used in this assessment is based on the assumption that exposure to PCBs and methylmercury is only occurring through the fish consumption pathway. Based on experience at similar sites, risks associated with other environmental media (e.g., groundwater, soil, air, surface water) are expected to be minor compared to fish consumption. However, if exposures from other media are significant, the HQs presented in this assessment may underestimate the effect of exposure from all media. If, following the suggestion of MOE (2005a), we assume that only 20% of the total exposure is from fish consumption, the HQs would be five times higher. Since there is no evidence suggesting that significant exposure through other pathways is occurring, this would be an overly conservative estimate of hazard.

The HQs for PCBs and methylmercury were not summed in this assessment because they do not share a common mechanism of action for most toxicity endpoints. However, there is evidence that PCBs and methylmercury may affect some of the same toxicity targets in an additive or greater than additive way. The ATSDR performed an extensive review of the literature concerning the potential for joint toxic action by PCBs and methylmercury consumed in fish (ATSDR 2004b) and in breast milk (ATSDR 2004a). While current scientific understanding is limited, some studies have shown joint toxic effects from PCBs and mercury exposure through fish and breast milk consumption for certain toxicity endpoints, the most significant being neurological effects. As summarized by ATSDR, there is evidence of synergism (i.e., greater than additive interaction) between PCBs and methylmercury in disrupting regulation of brain levels of dopamine that may influence neurological function and development. There is also evidence of additive joint action (i.e., no interaction) for a metabolic disorder (hepatic porphyria). For other toxic effects, there are no pertinent data available.

Since neurological effects are among the most sensitive effects produced by PCBs and methylmercury, the HQ estimates for each chemical developed for this screening level HHRA may underestimate the joint toxic action for this endpoint. Summing the HQs for each chemical would give a simple estimate of joint toxic effects, although it neglects potential synergistic interactions. A more detailed approach to evaluating the combined adverse effect from PCBs and methylmercury would be to screen individual toxicity target systems or organs separately using target-organ toxicity reference doses, such as those provided in ASTDR (2004b). Because the HQs for PCBs in the AOC are approximately equal to the target HQ of 1, the conclusions of this assessment might change if HQ were summed or if target-organ HQs were evaluated separately.

7 RISK MANAGEMENT

The foregoing analysis concluded that, under current conditions, mercury and, to a lesser extent, PCBs have the potential to cause adverse effects in fish populations, while PCBs (but not mercury) have the potential to cause adverse effects in individual mink and other piscivorous mammals, as well as in sport anglers and their household members. Concentrations of both mercury and PCBs in invertebrate and fish tissue exceed TRGs and local background concentrations. In light of these findings, management actions may be warranted to mitigate risks. In order to aid in the selection of appropriate management actions, this section derives sediment management goals based on risk, TRGs, background fish tissue concentrations, and source control of hot spots. The area and volume of sediment in JC and RPH exceeding sediment goals under these four scenarios are calculated and mapped. The degree of risk reduction and residual risk remaining following each of four remediation scenarios are also presented.

7.1 Sediment Management Goals

Sediment management goals are concentrations of mercury and PCBs in sediment that warrant consideration for sediment management decisions. Sediment management goals were developed based on four types of management criteria: 1) risk-based management goals; 2) guideline-based management goals; 3) background-based management goals; and 4) source control of hot spots. The first set, risk-based sediment management goals, was based on the outcome of the foregoing risk assessment. As such, the risk-based sediment management goals are most pertinent. Those ecological and human receptors and scenarios found to drive risks were used to back-calculate fish tissue concentrations that would not be expected to cause adverse effects in key receptors (i.e., risk-based target tissue concentrations). A food web bioaccumulation model based on the work of Grapentine et al. (2005) and Hope (2003) was then used to estimate sediment concentrations that would be required to attain those risk-based target fish tissue concentrations (i.e., sediment management goals). That model is described in detail in Appendix E.

The second set of goals, guideline-based management goals, was similarly estimated using the food web model, although in this case, the model generated the sediment concentration that would be required to attain TRGs for methylmercury and PCBs. The third set of goals, background-based management goals, is the set of modeled sediment concentrations that would be required to attain fish tissue concentrations of methylmercury and PCBs in longnose sucker, lake trout, and lake whitefish consistent with local background concentrations (i.e., mercury concentration equal to 0.18 mg/kg and PCB concentration equal to 0.40 mg/kg). The fourth set of goals is based on hot spots of total mercury in sediment, defined by fixed concentrations of total mercury in sediment. The derivation of each of these four sets of sediment management goals is detailed in below.

7.1.1 Risk-Based Sediment Management Goals

The foregoing risk assessment concluded that ecological receptors potentially at risk are fish (as a result of tissue concentrations of methylmercury and, in the case of longnose sucker, PCBs) and mink and other piscivorous mammals (as a result of consumption of fish containing PCBs). Bald eagles and other piscivorous raptors are only predicted to be at risk if they derive 75% to 100% of their prey from JC, an unlikely scenario given the relative areas of JC and RPH. All of the angler scenarios are predicted to be at risk from consumption of total PCBs in fish, except for adolescents and adults consuming fish from the entire AOC or JC. In contrast, none of the angler scenarios are predicted to be at risk from consumption of methylmercury in fish. The adult sport angler who fishes throughout the AOC is judged the most plausible and appropriately conservative of the angler scenarios. This scenario is used to develop sediment management goals protective of sport anglers.

The target concentration of methylmercury in fish tissue that is protective of fish is set equal to the TRV used in the ERA for that receptor, 0.20 mg/kg. The basis for that TRV is detailed in Section 4.4 above. The target concentration of PCBs in fish tissue that is protective of mink was calculated by solving the equation for DI for the concentration term and substituting the TRV for the DI term:

Eqn. 9

$$Ci = \frac{(TRV \times BW)}{\sum FIR \times \sum Pi} \times 10^3 \frac{g}{kg}$$

Where:

Ci	=	target fish tissue concentration (mg/kg)
TRV	=	toxicity reference value (mg/kg-day)
BW	=	body weight (kg)
FIR	=	food ingestion rate (kg/day)
Pi	=	proportion of diet composed of fish

The bases for these assumptions are detailed in Sections 3.3.1 and 4.5.5 above. The resultant target fish tissue concentration for PCBs based on protection of mink is 0.46 mg/kg.

Target fish tissue concentrations protective of sport anglers were calculated by solving the equation for EDI for the concentration term and substituting the TDI for the EDI term:

Eqn. 10

$$Ci = \frac{BW \times AT \times TDI}{CL \times FCR \times FI \times EF \times ED} \times 10^3 \frac{g}{kg}$$

Where:

AT	=	averaging time (days)
TDI	=	tolerable daily intake (mg/kg-day)

CL	=	cooking loss factor (unitless)
FCR	=	fish consumption rate (g/day)
FI	=	fraction ingested from source (unitless)
EF	=	exposure frequency (day/year)
ED	=	exposure duration (year)

The bases for all assumptions related to anglers are detailed in Section 6 above. Although the screening level HHRA predicted potential adverse health effects for many of the angler scenarios, the most probable yet appropriately conservative of the scenarios—adult sport anglers that consume fish from throughout the AOC—is the focus of this analysis. As shown in Table 6-5, such anglers are not predicted to be at risk from current concentrations of either methylmercury or PCBs in fish, although pre-adult populations may be at risk from PCBs. The target PCB concentration in fish protective of this scenario is 0.58 mg/kg.

Risk-based sediment management goals were then calculated from these target fish tissue concentrations using a food web model developed from the work of Grapentine et al. (2005) and Hope (2003), as described in Appendix E. In brief, the model is composed of 14 compartments representing the physical and biological properties of the Peninsula Harbour system. The compartments include: 1) source media (sediment and surface water); 2) primary consumers; 3) secondary consumers; and 4) tertiary consumers. The model employs chemical-specific bioconcentration factors (BCFs), biota sediment accumulation factors (BSAFs), and biomagnification factors (BMFs) that describe the movement of COCs through the system and into fish tissue. The BMFs are adjusted based on bodyweight and age of fish, using the fraction of equilibrium attained at the time of consumption (f_E) (Hope 2003). This adjustment accounts for the changes in tissue concentrations that occur as a fish grows. Tissue concentrations were modeled based on methylmercury and lipid-normalized PCBs. Resultant risk-based sediment management goals for methylmercury and total PCBs are listed in Tables 7-1 and 7-2, respectively.

7.1.2 Guideline-Based Sediment Management Goals

Derivation of guideline-based sediment management goals follows a similar procedure to that described above for risk-based sediment management goals, except that the TRG for methylmercury and the extrapolated TRG for total PCBs were used instead of the risk-based target fish tissue concentrations. For methylmercury, the TRG of 0.092 mg/kg is used, as discussed in Section 4.5.6.

As also discussed in Section 4.5.6, Environment Canada's (1998) TRG for TEQs is equal to 0.79 ng/kg (Environment Canada 1998). There were insufficient data to develop a sediment management goal based on the TEQ TRG. Samples collected from Peninsula Harbour and nearby regions of Lake Superior as part of the SFCMP were used to develop a data set of 25 paired TEQ and total PCB results. These paired results were used to characterize the relationship between TEQ and total PCB concentrations in local fish. A total PCB concentration of 0.13 mg/kg is extrapolated from the TRG for TEQs of 0.79 ng/kg.

Guideline-based sediment management goals were then calculated using the same food web model described above. Resultant guideline-based sediment management goals for methylmercury and total PCBs are listed in Tables 7-1 and 7-2, respectively.

7.1.3 Background-Based Sediment Management Goals

Derivation of background-based sediment management goals followed a similar procedure to that described above, except that the average of extrapolated whole body fish tissue concentrations from the SFCMP Zone 7 were used instead of the risk-based target fish tissue concentrations or the TRG. Thus, the assumed background concentration of methylmercury in fish (averaged across longnose sucker, lake trout, and lake whitefish) is 0.18 mg/kg, while the assumed background concentration of PCBs in fish is 0.40 mg/kg. Resultant background-based sediment management goals for methylmercury and total PCBs are listed in Tables 7-1 and 7-2, respectively.

7.1.4 Hot Spot-Based Management Goals

Hot spot-based sediment management goals were determined subjectively based on the distribution of concentrations of total mercury in surface sediment and the locations of elevated concentrations. A similar hot spot analysis for PCBs was not conducted due to excessive uncertainty in the spatial distribution of elevated total PCB concentrations in RPH. The objective of the total mercury hot spot analysis is to identify those portions of JC with the most elevated concentrations and the area and volume of sediment associated with them, so that a management scenario associated with source control may be considered. As shown in Table 7-3, the hot spot-based management goals considered were 2 mg/kg, 3 mg/kg, 6 mg/kg, and 10 mg/kg total mercury in sediment. The hot spot is defined by concentrations of total mercury, rather than methylmercury, because total mercury acts as a source for methylmercury generation and because rapid turnaround analysis and field screening measurements (which are necessary during remedial activities) are more feasible for total mercury than for methylmercury. Additionally, there is substantial overlap in the distribution of elevated concentrations of total mercury, methylmercury, and PCBs in JC sediment.

7.2 Management Areas and Volumes

In order to characterize the area and volume of sediment potentially warranting management under the four sets of sediment management goals developed above, the spatial distribution of mercury and PCBs in JC and RPH was first considered under current conditions. Simulations of various remedial scenarios were then employed to determine the area and volume of sediment warranting remediation in order to attain the sediment management goals in JC and RPH. The specific methods and findings are described below.

7.2.1 Spatially Weighted Average Concentrations

Because both the fish and the humans and wildlife that feed on them are mobile, actual exposures to mercury and PCBs in sediment are effectively averaged over space. Additionally, it is reasonable to assume that it is the surficial sediment, as opposed to deeper sediment, that most strongly influences fish exposure. Therefore, it is appropriate to compare SWACs of COCs in surface sediment (0 cm to 5 cm and 0 cm to10 cm) of JC and RPH to the sediment management goals. SWACs are calculated using Thiessen polygons (Davis 1986, as cited in DOE 2006). Thiessen polygons are derived from a set of sample location points. They bound an area in which any given location is nearest to the associated sample point relative to all other sample points. They are formed by the perpendicular bisectors of the lines between sample locations. SWACs are calculated as:

Eqn. 11

$$SWAC = \frac{\sum Ci \times Ai}{\sum Ai}$$

Where:

SWAC = spatially weighted average concentration (mg/kg)

Ci = concentration of COC in polygon i (mg/kg)

Ai = area of polygon i (ha)

Figure 7-1 illustrates the Thiessen polygons used to calculate the SWAC of methylmercury in surface sediment (0 cm to 5 cm and 0 cm to10 cm) under current conditions. The SWAC of methylmercury in the entire AOC is 0.0029 mg/kg, while the SWACs of methylmercury in JC and RPH are 0.0051 mg/kg and 0.0019 mg/kg, respectively. That is, the concentrations of methylmercury in surface sediment are elevated in JC relative to RPH. Table 7-1 compares these current SWACs to the sediment management goals developed above, illustrating that the current SWAC of methylmercury in RPH is less than both the risk-based sediment management goal that is protective of effects in fish and the background-based sediment management goal. In contrast, the current SWAC of methylmercury in JC is greater than all three sediment management goals listed. The SWAC of methylmercury in JC is 2.6-fold greater than the risk-based sediment management goal.

Figure 7-2 illustrates the Thiessen polygons used to calculate the SWACs of total PCBs in sediment under current conditions. The SWAC of total PCBs in the entire AOC is 0.11 mg/kg, while the SWACs of total PCBs in JC and RPH are 0.14 mg/kg and 0.12 mg/kg, respectively. That is, concentrations of PCBs in sediment are similarly elevated in JC and RPH. Concentrations of total PCBs in JC and RPH sediment are not significantly different based on the Wilcoxon rank sum test ($p = 0.49$). Table 7-2 compares these current SWACs to the sediment management goals derived above, illustrating that the current concentrations of PCBs in surface sediment in both JC and RPH are greater than risk-based management goal protective of mink, the guideline-based management goal, and the background-based management goal. The current concentration of PCBs in JC exceeds the mink sediment management goals by 2.3-fold. The current concentration of PCBs in RPH exceeds the mink sediment management goals by 2-fold.

7.2.2 Management Areas

As shown in Tables 7-1 and 7-2, current SWACs exceed many of the sediment management goals based on risk, guidelines, and background. In order to determine the areas of sediment warranting management based on these three types of sediment management goals, the highest measured concentrations of methylmercury and total PCBs in surface sediment were sequentially replaced with concentrations of 0 mg/kg. SWACs were repeatedly recalculated until they were found to be equal to or less than each sediment management goal. The substitution of non-zero measured sediment concentrations with 0 mg/kg is analogous to implementation of a fully effective sediment management strategy within the specified area.

The sum of the areas of the polygons with substituted concentrations of 0 mg/kg represents the predicted area of sediment theoretically warranting management for a given management scenario ("management area"). In selecting management areas, contiguous management areas were chosen preferentially over discontinuous areas. It should be noted that there are multiple possible ways to achieve a given SWAC while achieving the main objective of the exercise (management of highest concentration areas, maximizing continuity). Consequently, the selected management areas are examples that are subject to modification for reasons of feasibility and/or as additional data become available. Low sample density in RPH results in very large areas being characterized based on a single sample result.

Management scenarios were developed for those management goals that are exceeded by current SWACs. Table 7-3 lists the area (in m^2) warranting management to achieve the hot spot-based sediment management goals. Table 7-4 lists the area warranting management to achieve the risk-based, guideline-based and background-based sediment management goals. Management areas associated with each type of sediment management goal are mapped in Figures 7-3 through 7-10. Findings by scenario are summarized below.

7.2.2.1 Risk-Based Management Areas

Risk-based management areas were evaluated for both ecological and human receptors, as discussed in the following two subsections.

7.2.2.1.1 Ecological Risk

When focusing on risks to fish, management actions are warranted to mitigate methylmercury risks if fish are assumed to obtain 25%, 50%, 75%, or 100% of their prey from JC (Figure 7-3). As discussed in Section 3.2, the assumption that fish derive 25% of their diet from JC is a plausible and appropriately conservative scenario. If fish are assumed to obtain 100% of their prey from RPH, management actions are not required to mitigate methylmercury risks (Figure 7-3).

As detailed in Section 5.3 above, mink are predicted to be at risk from concentrations of PCBs – but not methylmercury – in their aquatic prey. HQs for mink from PCBs range from 1 to 9 based on average dietary concentration and depending upon relative proportions of prey derived from

JC and RPH. Management actions required to mitigate risks from PCBs to mink under the five AUF scenarios are illustrated in Figure 7-4.

7.2.2.1.2 Human Health Risk

The sediment management goal protective of sport anglers (0.19 mg/kg) is not exceeded by the SWAC in either JC or RPH. Thus, no sediment management is required to mitigate risks from PCBs to sport anglers. This determination appears to conflict with the prediction of risk to some sport angler age groups based on measured fish concentrations presented in Section 6. However, this evaluation of sediment concentrations is based on comparing the sediment management goal to average sediment concentrations. In contrast, the analysis in Section 6 is based on risk estimates calculated using the 95% UCLM, a conservative estimate of the average. If the risks estimated from fish tissue concentrations are based on average concentrations, no risk to any of the sport angler age groups is predicted. Thus, the analysis of sediment concentrations presented here is, in fact, consistent with analysis based on fish concentrations presented in Section 6.

7.2.2.2 Guideline-Based Management Areas

Although the management areas based on ecological risk (discussed in Section 7.2.2.1.1 above) and those based on TRGs are both based on protection of piscivorous wildlife, the guideline-based sediment management goals are considerably lower than the ecological risk-based sediment management goals; consequently the guideline-based management areas are considerably larger than the ecological risk-based management areas. This difference in goals and management areas reflects the more conservative ecotoxicological assumptions employed in the derivation of TRGs, relative to the ERA, as previously discussed in Section 5.3. As illustrated in Figures 7-6 and 7-7, extensive management of sediment in JC and RPH is necessary to achieve the TRG for methylmercury or the extrapolated TRG for total PCBs in fish.

7.2.2.3 Background-Based Management Areas

Figure 7-8 illustrates that the current SWAC of methylmercury in RPH sediment is predicted to result in fish tissue concentrations of methylmercury that are consistent with background, when averaged across longnose sucker, lake trout, and lake whitefish, but that management of JC sediment would be necessary in order to achieve background concentrations in fish that derive their entire diet from JC. Figure 7-9 illustrates that extensive management of JC and RPH sediment would be warranted to achieve background concentrations of total PCBs in fish tissue.

7.2.2.4 Hot Spot-Based Management Areas

The foregoing analyses indicate that management of sediment to achieve risk-based, guideline-based, or background-based management goals would require extensive remedial actions in both JC and RPH. The existing hot spot of mercury and PCBs in surface sediment of JC serves as an ongoing source of contamination to the rest of JC and RPH. Therefore, hot spot management for purposes of source control warrants consideration. As shown in Figure 7-10, management areas of 25 ha, 18 ha, 8 ha, and 4 ha in JC are associated with the hot spots defined by concentrations of total mercury in sediment greater than 2 mg/kg, 3 mg/kg, 6 mg/kg, and 10 mg/kg, respectively. Geospatial interpretation of these hot spots is shown in Figure 7-11. The overlap of the total mercury hot spots and methylmercury in JC is illustrated in Figure 7-12.

In order to depict the spatial distribution of mercury hot spots, a geospatial interpretive tool called inverse distance weighting was used to interpolate sediment concentrations in areas lacking sampling data. This tool estimates concentrations in unsampled areas, based on the concentrations measured elsewhere, weighting those samples closest to each interpolated point more heavily than those that are more distant. In applying this tool, there is a trade-off between the relative weight assigned to more distant points and the degree of "structure" in the output (more weight assigned to distant points results in a smoother representation). Thus, to smooth the concentration isopleths, we increased the weight assigned to distant points. However, this practice creates some areas where high concentrations some distance away influence sparsely sampled areas, as in the case of the southwestern shore of Jellicoe Cove. As with any interpretive geospatial tool, there is uncertainty in the interpolated concentrations at unsampled locations.

The overlap of total mercury and methylmercury throughout the entire AOC is illustrated in Figure 7-13. When Figures 7-12 and 7-13 are compared to the distribution of TOC shown in Figure 1-4, it appears that mercury methylation is influenced by TOC, as expected. In other words, locations with elevated methylmercury concentrations in sediment correlate with areas with elevated total mercury and TOC in sediment.

In order to help identify the total mercury concentration in sediment that is most appropriate for defining hot spots, a hockey stick (or piecewise) regression analysis was conducted. As illustrated in Figure 7-14, the relationship between the area remediated and the residual concentration of mercury in the sediment trends toward vertical at low concentrations (i.e., below 3.2 mg/kg) and trends toward horizontal at higher concentrations. That is, as the area of the hypothetical remedial footprint increases, there is at first a rapid reduction in the residual mercury concentration. As the footprint increases further, however, the rate of reduction is dramatically reduced. The initial, rapid reduction in residual concentrations is consistent with sediment in a hot spot, while the slow subsequent reduction is consistent with regional concentrations. Given that there appear to be two discrete populations of mercury-contaminated sediment, a hockey stick regression was used to determine the threshold concentration of mercury, below which additional removal will have diminishing effect on residual concentrations of mercury. The threshold concentration is defined as the intersection between two linear regressions that describe the areas with hot spots and regional contamination. A threshold concentration of 3.2 mg/kg was calculated by optimizing its value to minimize the residual sums of squared of the two regressions using custom code in R (R Development Team 2006). The predicted remedial footprint of 15 ha is associated with the threshold concentration of 3.2 mg/kg. This finding agrees well with the management areas developed using Thiessen polygons (i.e., 18 ha associated with the hot spots defined by surface sediment concentrations of total mercury greater than 3 mg/kg). Given the high R^2 value

(0.985) and statistical significance ($p < 0.001$) of the regression analysis, such good agreement between the two methods is not surprising.

7.2.3 Management Volumes

Among other factors, sediment management decisions will likely consider the volume of sediment warranting management, given that the cost of one remedial option (dredging) depends upon the volume of sediment remediated. Towards that end, the available data on sediment depths within the AOC were used to map estimated depths of contaminated sediment in JC. The most comprehensive study of sediment depths to date was performed by BEAK (2000). Cores completely penetrating the recent potentially-contaminated sediment and extending into the underlying pre-industrial sediment were advanced at 48 locations arranged on a grid. The pre-industrial sediment, identified as glacial clay and outwash, were shown to be relatively uncontaminated by COCs (BEAK 2000). Based on an analysis of the stratigraphy presented in core logs, the vertical extent of recent sediment was estimated at each core location and interpolated throughout JC using kriging (BEAK 2000). The contour map of sediment depths for mercury concentrations greater than 0.2 mg/kg presented in Figure 23 of BEAK (2000) was digitized and the average sediment depth within the well-characterized portion of JC was calculated using a Geographic Information System (GIS). Depths of cores advanced to refusal by other researchers were also considered as available. According to data obtained from BEAK (2000), the depth of sediment with mercury concentrations greater than 0.2 mg/kg in the characterized portion of JC averages 15 cm and ranges from 1.6 cm to 42 cm. However, certain cores collected in JC for other studies reached a depth of 54 cm. Figure 7-15 maps contaminated sediment depths in JC.

For purposes of conservatively calculating volumes of sediment warranting management to achieve risk-based, guideline-based, and background-based sediment management goals within JC, the maximum depth of 54 cm was applied as an upper bound estimate. The average of 15 cm was applied as a central tendency estimate of sediment depth in JC. Based on very sparse sediment depth data for RPH, average and maximum estimates of 7.6 cm and 90 cm, respectively were used to estimate volumes of sediment warranting management in RPH to achieve risk-based, guideline-based, and background-based sediment management goals.

As shown in Table 7-4, the volumes of sediment warranting management under the risk-based, guideline-based, and background-based management scenarios vary between zero and millions of cubic yards. Based on the “rule of thumb” cost of dredging of approximately \$100 to \$300 per m^3 (NRC 2001), this analysis suggests that remediation to achieve acceptable risks, guidelines, and/or background would cost hundreds of millions to billions of dollars. Furthermore, technical and logistical constraints (e.g., water depth) may preclude certain types of management actions in RPH.

Given these circumstances and ongoing releases of mercury and PCBs from the JC hot spot to the rest of JC and RPH, the hot spot-based sediment management goals warrant further consideration, in order to achieve the objective of source control. The apparent spread of methylmercury and PCBs from the JC hot spot is visible in Figures 1-6 through 1-8, particularly in light of the bathymetry (Figure 1-2) and distribution of TOC (Figure 1-4). Location-specific sediment depth information (from BEAK 2000) was used to estimate the average depth of sediment in the three hot spot areas. Resultant volumes of sediment associated with hot spot sediment management actions listed in Table 7-3 for sediment management goals of 2 mg/kg, 3 mg/kg, 6 mg/kg, and 10 mg/kg range from approximately 8,000 m^3 to 43,000 m^3 (i.e., remediation costs of approximately $780,000 to $13,000,000). The choice of one hot spot-based sediment management goal (either from these three options or any other value) is essentially a management decision that balances cost with desired outcome, relative risk reduction, and acceptability to stakeholders. However, the 3 mg/kg hot spot definition is most consistent with the threshold concentration identified by the hockey stick regression analysis. Thus, there are diminishing returns for source removal of concentrations less than 3 mg/kg.

It is worth noting that this assessment and quantification of management volumes is largely theoretical in that it does not account for the many considerations that are critical to remediation decisions. The next stage of analysis—the sediment management options analysis—evaluates a full range of remedial options in depth. More accurate and complete costs associated with the various alternatives are also provided in the sediment management options analysis.

7.3 Residual Risks and Risk Reduction under Hot Spot Scenarios

Although the primary objective of hot spot remediation is source control, such a management strategy provides the collateral benefit of reducing risks simply through control of sediment with elevated concentrations of mercury and total PCBs. This subsection quantifies the relative reduction in risks associated with the four hot spot scenarios (i.e., hot spots defined by total mercury in sediment exceeding 2 mg/kg, 3 mg/kg, 6 mg/kg, and 10 mg/kg). In addition, residual HQs following hot spot remediation are calculated. This analysis focuses on the two receptors judged both plausible and likely to be adversely affected by current concentrations of mercury and PCBs in JC, namely fish (from methylmercury) and mink (from total PCBs).

As a first step in these calculations, the food web model was used to predict mean concentrations of methylmercury and total PCBs in fish under current (baseline) conditions and four post-hot spot remediation scenarios (i.e., 2 mg/kg, 3 mg/kg, 6 mg/kg, and 10 mg/kg) (Table 7-5). Baseline and current EPCs vary across receptors. In particular, whole body 45-cm longnose sucker concentrations are used to evaluate baseline and post-remediation risks to fish, because longnose sucker is the fish species predicted to be at highest risk. Thus, conclusions for longnose sucker are protective of all other fish species (e.g., lake trout, walleye, lake whitefish). Whole body 20-cm longnose sucker, lake trout, and lake whitefish (averaged across species) concentrations are used to evaluate baseline and post-remediation risks to mink. Based on the baseline and post-remediation EPCs shown in Table 7-5, post-remediation HQs and percent reduction in HQs for each receptor (i.e., longnose sucker, mink) are calculated in Tables ,7-6, 7-7 and 7-8, as discussed in the following subsections.

7.3.1 Fish

Methylmercury concentrations in fish predicted for JC under baseline conditions and under four hot spot remediation scenarios were compared to the TRV of 0.20 mg/kg, to assess residual risks and risk reduction for fish. As shown in Table 7-6, hot spot remediation is predicted to reduce HQs for fish from methylmercury by 7% to 62%, depending on the mercury concentration used to define the hot spot and the fraction of time that fish are assumed to forage in JC and RPH. If the hot spot is defined by total mercury in sediment greater than either 2 mg/kg or 3 mg/kg, residual HQs for fish are not predicted to exceed the target HQ of 1 regardless of the amount of time they forage in JC and RPH. If the hot spot is defined by total mercury in sediment greater than 6 mg/kg, residual HQs for fish are not predicted to exceed the target HQ of 1 if they forage in JC 25% or 50% of the time. If the hot spot is defined by total mercury in sediment greater than 10 mg/kg, residual HQs for fish are not predicted to exceed the target HQ of 1 they forage in JC 25% of the time. In summary, remediation of the hot spot defined by either 2 mg/kg or 3 mg/kg is predicted to mitigate the current risks posed by mercury to sportfish and bottom-dwelling fish species.

7.3.2 Mink

Residual HQs for mink were calculated based on PCB concentrations predicted for JC under baseline conditions and under four hot spot remediation scenarios (Table 7-7). As shown in Table 7-7 and as expected, the degree of risk reduction increases with increasing extent of remediation. Under the most aggressive hot spot remedy (hot spot defined by 2 mg/kg), a 44% reduction in risk is achieved for those mink that forage 75% of their time in JC. At the other extreme, a 4% reduction in risk is achieved if the hot spot is defined by 10 mg/kg and mink are assumed to forage 25% in JC and 75% in RPH. All hot spot scenarios result in residual HQs slightly greater than 1, (i.e., range from 1.0 to 1.7). When rounded to one digit as is customary for HQs, remediation of the hot spot defined by either 2 mg/kg or 3 mg/kg is predicted to achieve the target HQ of 1.

In summary, remediation of the hot spot defined by either 2 mg/kg or 3 mg/kg is predicted to result in post-remediation HQs for fish and mink that are generally at the target level of 1. Given the smaller area of the hot spot defined by 3 mg/kg, it would be more cost-effective to clean up to this target, since it is predicted to yield an essentially equivalent residual HQ as that yielded by clean up to 2 mg/kg. As previously discussed, 3 mg/kg is also essentially equal to the point of inflection generated by the hockey stick regression (i.e., clean up to 3 mg/kg is considerably more cost-effective than cleanup to any lower level. Remediation to any higher level than 3 mg/kg would yield residual HQs that round to 2 (i.e. above the target hazard level). Remediation of sediment containing concentrations of total mercury equal to or greater than 3 mg/kg would cut the concentration of methylmercury in JC in half, from 0.0052 mg/kg to 0.0027 mg/kg.

7.4 Uncertainty

Because uncertainty is an unavoidable element of all risk assessments, it is also unavoidable in analyses that rely on risk assessment, such as this one. Sources of uncertainty in this analysis are summarized below. In addition, Appendix E describes a probabilistic sensitivity analysis conducted on the food web model.

- *Characterization of methylmercury and PCBs in RPH sediment.* The limited number of surficial sediment samples analyzed for methylmercury and PCBs in RPH contributed uncertainty to the calculation of management areas. The spatial distribution of methylmercury and PCBs in RPH is poorly characterized. Based on the spatial distribution of methylmercury data (as presented in Figure 7-1), it is possible that methylmercury concentrations are slightly underestimated in RPH due to the lack of samples near JC. Conversely, PCB concentrations are likely overestimated due to the lack of samples available to define hot spots.
- *Calculation of sediment volume in RPH.* Due to a lack of data on sediment depths in RPH, the average and maximum depths from the available cores were used to calculate a range of volumes warranting management. The limited number of core samples in RPH, coupled with the large area of RPH, contributes to uncertainty in this calculation and likely leads to an overestimation of the volume of sediment warranting management in RPH.
- *Calculation of sediment volume in JC.* Sediment depth data for JC were obtained from two sources: contaminated sediment depth contours presented in BEAK (2000) and maximum depth of available sediment cores. While similar, these two sources of data do not always agree. In order to produce a conservative estimate of sediment depth, the greater of the values generated from the two sources was used. As such, the average sediment depth (15 cm) was obtained from BEAK (2000), while the maximum sediment depth (56 cm) was obtained from the core end depth. This practice likely leads to an overestimation of sediment volume warranting management.
- *Determination of Thiessen polygons.* Due to the spatial distribution of methylmercury and PCB sample points within the AOC, separate sets of Thiessen polygons were created for JC and RPH. The segregation of JC and RPH polygons likely created a more accurate polygon distribution within JC and RPH; however, it makes the simplifying assumption that there is no interconnection or influence between the two areas with respect to sediment transport or water flow, when clearly that is not the case.
- *Calculation of areas warranting management.* In order to determine the areas of sediment warranting management under various scenarios, the concentrations in selected Thiessen polygons were set to zero until the SWAC was less than the sediment management goal. Many different combinations of polygons could be selected to achieve a given sediment management goal. Contiguous polygons with the highest concentrations were chosen preferentially until the goal was met. In many cases, alternate management scenarios could be created to achieve the same goal.

- *Remedy effectiveness.* The assumption that management actions are 100% effective (i.e., result in 0 mg/kg concentrations) is not likely accurate for certain remedies, including dredging and in situ enhanced bioremediation. Indeed, Patmont and Palermo (2007) estimate that generated residuals resulting from dredging typically range from 2% to 9%. Thus, management areas and volumes may be underestimated if a dredging remedy is selected for this site.
- *Risk estimates for mink versus sediment management*. Although mink are not predicted to be at risk to PCBs in fish tissue if 100% of their diet is obtained in the JC, the model predicts that some sediment warrants management under Scenario 1 (i.e., 100% JC). This is likely due to uncertainty in the relationship between sediment and fish tissue concentrations.
- *Risk estimates for anglers versus sediment management.* As described in Section 6, risk estimates based on the 95% UCLM of mean fish tissues concentrations indicate a potential risk to sport anglers from PCBs. However, PCB SWACs in JC and RPH are below the sediment management goal protective of sport anglers. This difference is due to use of the 95% UCLM concentrations in the screening level HHRA in Section 6 in order to provide a conservative estimate of risk. If the risk to sport anglers is estimated using average fish concentrations, there is no predicted risk of adverse health effects. The actual fish concentrations being consumed by sport anglers are uncertain, resulting in uncertainty in estimates of risk.

As detailed in Appendix E, a probabilistic sensitivity analysis was conducted to evaluate the food web bioaccumulation model's sensitivity to the choice of input parameters. Those model parameters that could have been influenced by receptor size and/or the accuracy of the literature values and calibrated values were assigned a distribution of values. The distributions encompassed ranges of likely values for the parameters of interest. The distributions of most parameters were described as triangular, based on the minimum, maximum, and most likely values. Crystal Ball (Decisioneering, Inc., Denver, Colorado) was used to repeatedly execute the model as a Monte Carlo simulation. A total of 10,000 iterations were performed, each time based on values selected from within the assigned range, based on the probability distribution. The resulting sediment management goals were compiled. The sensitivity analysis demonstrates that sediment management goals associated with the 10th and 90th percentiles generally span a factor of two or three and that 50th percentile values are generally consistent with the point estimates generated from best estimates of the model parameters. The relatively small range of predicted concentrations from the sensitivity analysis shows that the model is robust and is relatively insensitive to changes in the input parameter values. Thus, the model predicts sediment management goals with acceptable accuracy.

8 SUMMARY

This environmental risk assessment for Peninsula Harbour includes both an ecological risk assessment (ERA) focused on risks posed to fish and wildlife and a screening level human health risk assessment (HHRA) focused on risks posed to people who eat fish caught in Peninsula Harbour. It also supports the development of a sediment management strategy for Peninsula Harbour. Objectives of this report were to: 1) estimate risk posed by mercury and PCBs in Peninsula Harbour sediment and biota to human anglers and ecological receptor species; 2) develop numerical sediment management goals based on the findings of the risk assessment, existing guidelines, background concentrations in fish, and source control of hot spots; 3) estimate the area and volume of sediment requiring management in order to achieve those goals; and 4) predict the residual risks that would remain following several different management scenarios. Because the majority of analytical data available for Peninsula Harbour pertain to JC and in order to ensure that the findings aid effective risk management decisions for JC, the AOC was divided into two discrete areas for purposes of this ERA: JC and RPH. That division of the harbour is a management decision; there are no physical barriers that impede water flow, sediment transport, or human or biota movement between JC and RPH.

The ERA evaluated the following assessment endpoints: 1) community structure, survival, and reproduction of benthic invertebrates; 2) survival and reproduction of fish; 3) survival and reproduction of piscivorous birds; and 4) survival and reproduction of piscivorous mammals.

Measurement endpoints used to evaluate potential risks to benthic invertebrates were: 1) abundance, richness, and diversity of assemblages, relative to reference stations of comparable habitat characteristics; 2) sediment toxicity to multiple invertebrate species, as measured in laboratory toxicity tests; and 3) concentrations of mercury and PCBs in sediment in relation to appropriate sediment quality guidelines and concentrations reported in literature to be harmful to benthos.

Measurement endpoints used to evaluate potential risks to fish were: 1) concentrations of mercury and PCBs in tissues of representative species in relation to concentrations reported in literature to be harmful to fish; 2) fish community structure and recruitment relative to reference area(s); and 3) comparison of mercury and PCB concentrations in fish from the AOC to concentrations in fish from a local Lake Superior reference area (Zone 7).

Measurement endpoints used to evaluate risks to piscivorous wildlife were: 1) comparison of modeled dietary intake of mercury and PCBs by two representative avian species (common loons and bald eagles) and one representative mammalian species (mink) to doses reported in the literature as thresholds for adverse effects on survival or reproduction (i.e., HQs); and 2) comparison of species-specific and location-specific whole body fish and invertebrate tissue concentrations of mercury and PCBs to TRGs.

Effects on the benthic invertebrate community were evaluated based on the site-specific relationship between sediment chemistry, toxicity, and benthic community composition (Milani et al. 2002). Benthic invertebrate community composition in JC was found to be different than in reference sites. The observed differences were attributed to organic enrichment and water depth, rather than chemical contamination. In subchronic and chronic laboratory tests with multiple species, very few sediment samples from the AOC exhibited any indication of toxicity. In most cases where toxicity was indicated, the observed effects were attributable to physical characteristics of the sediment. No relationship was observed between mercury concentrations and sediment toxicity; PCBs were not measured in the toxicity study. Concentrations of mercury in sediment generally exceeded low end empirical screening values. The high-end empirical screening values for total PCBs were infrequently exceeded and Ontario's high-end screening value (the SEL) for mercury was not exceeded in any RPH sediment samples. Observed concentrations are similar to levels that have been found to be nontoxic at some other mercury-contaminated sites (Sferra et al. 1999). Similarly, total PCB concentrations in sediment throughout the AOC are below levels at which PCBs are likely to be a primary cause of toxicity to benthic invertebrates (Fuchsman et al. 2006). Taken together, these lines of evidence indicate that benthic invertebrates in the AOC are not at significant risk due to mercury or PCBs in sediment.

To evaluate risks posed to fish, fish tissue concentrations were compared to TRVs for methylmercury and total PCBs. The mean HQ for longnose sucker collected from RPH exceeds the target HQ of 1 (HQ=2). Because the mean HQ is a central tendency estimate, this finding suggests that adverse effects in longnose suckers from mercury may propagate to population-level effects. Although the mean HQs for mercury in no other species of fish exceed the target HQ of 1, the 95th percentile HQs for mercury exceed 1 for lake trout, walleye, and lake whitefish (as well as longnose sucker). Because the 95th percentile HQ is based on the most highly exposed individual fish, adverse effects from mercury are more likely to be limited to individuals than to propagate to population-level effects in sportfish species. Neither mean nor 95th percentile HQs for PCBs exceed 1 for any fish species. The 95th percentile HQs for longnose suckers exposed to PCBs are 2 in both the entire AOC and RPH, but is 0.4 in JC. Therefore, PCBs are not likely to cause adverse effects in other fish species at either the individual or population levels.

Risks to fish were also evaluated based on available data on the fish community. However, recent data characterizing fisheries resources in JC were not designed to identify the occurrence or lack of toxicity to fish due to COCs. Fish in JC were less abundant but more diverse than in Carden Cove. No longnose suckers were sampled for chemical analysis from either JC or Carden Cove, limiting the ability to draw comparisons between the biological and chemical characterization of the fish species at greatest risk. Although the fisheries survey results were not indicative of severe adverse effects, this measurement endpoint is inconclusive with regard to more subtle effects on the most highly exposed species.

For longnose sucker, lake trout, and lake whitefish combined, as well as for longnose sucker alone, average fillet and extrapolated whole body concentrations of mercury and PCBs are higher in Peninsula Harbour than in Zone 7 (i.e. background) of SFCMP. This trend is also apparent when median fillet concentrations are compared and when upper percentile fillet concentrations are compared. This comparison suggests that both regional sources (e.g., other point sources, atmospheric deposition) and site-specific sources (e.g., contaminated sediment in the AOC) contribute to mercury and PCB bioaccumulation in fish within the AOC.

In summary, comparisons of tissue concentrations to TRVs indicate unacceptable risks to sportfish (e.g., lake trout, walleye, lake whitefish) and bottom dwelling fish (e.g., longnose sucker) species in the AOC due to mercury and, in the case of longnose sucker, PCBs. Predicted risks are highest in longnose sucker, where reproductive impairment may propagate to population-level impacts. Although the potential for population-level impacts in sportfish is less clear, mercury is predicted to pose unacceptable risks in the most highly exposed individual lake trout, walleye, and lake whitefish. With the exception of longnose sucker, PCBs are not predicted to pose an unacceptable risk to fish. Available biological data are insufficient to confirm or refute conclusions based on comparisons of tissue concentrations to TRVs.

Risks to piscivorous wildlife were evaluated based on HQs and through comparison of concentrations of mercury and PCBs in prey tissue to TRGs. All calculated HQs for common loons are less than 1, regardless of whether the mean or 95% UCLM concentration is applied as the dietary concentration and regardless of the relative proportions of prey derived from JC and RPH. HQs for exposure of bald eagles to PCBs and mink to mercury are also consistently less than 1, regardless of the dietary concentration applied and the relative proportion of prey derived from JC and RPH. HQs for exposure of bald eagles to mercury based on the mean concentration do not exceed 1 for any exposure scenario. However, when the 95% UCLM is applied as the dietary concentration, HQs exceed 1 for bald eagles exposed to mercury under Scenarios 1 and 2. The HQ is less than 1 for bald eagles exposed to mercury under the most realistic, yet appropriately conservative scenario (Scenario 4). Thus, mercury in AOC fish is not predicted to pose an unacceptable risk to bald eagles. HQs for exposure of mink to PCBs range from 1 to 9, depending on the dietary concentration applied and the relative proportion of prey derived from each part of the AOC. Although HQs equal 1 for mink exposed to PCBs when 100% of prey is derived from JC, mink are not expected to derive all of their prey from JC due to habitat constraints. Individual mink are predicted to be adversely affected by total PCBs in prey, but not by methylmercury in prey. This conclusion is also supported by HQs for PCB homologues and TEQs, based on mink body burdens, which also consistently exceeded 1.

The majority of available prey tissue concentrations exceed TRGs. This finding holds regardless of the species evaluated and regardless of whether the mean or 95% UCLM concentration in tissue is compared to the TRG. Uncertainty is associated with both the HQ and TRG measurement endpoints, although greater conservatism is inherent in the TRG than in the HQ calculations. While the TRG comparisons are more conservative than the HQs, on balance, the HQs do not appear to underestimate risks to piscivorous wildlife. Thus, the ERA concludes that the only wildlife receptor expected to be at risk is the mink, due to exposure to PCBs but not mercury.

The screening level HHRA included a survey of residents of Marathon and the Pic River Reserve to determine residents' fishing and fish eating practices. Risks were evaluated based on: 1) results of the survey; 2) an understanding of what age groups of individuals might eat fish (i.e., toddlers, children, adolescents and adults); and 3) information on the concentrations of PCBs and mercury in fish from Peninsula Harbour.

Based on the fish consumption survey, there is no evidence of subsistence fishing by First Nations members in Peninsula Harbour. People who reported eating the most sport-caught fish tend to fish in locations other than Peninsula Harbour. Although an advisory on eating fish caught in Peninsula Harbour is currently in place for the Peninsula Harbour AOC, survey results suggest that the fish consumption advisory does not influence anglers' decisions about where to fish or whether to eat the fish that they've caught. While local people reported eating many different fish species, only one person reported eating longnose suckers. Therefore, longnose sucker tissue concentrations were not considered in the estimation of exposure of anglers or their family members. The fish consumption survey also indicated that 80% of those who responded to the survey eat less than 17.5 grams per day of sport-caught fish. This rate (17.5 grams per day) was therefore used in the screening level HHRA as a typical fish consumption rate for the area. Based on the survey results from anglers who fished at least some of the time in the Peninsula Harbour AOC and who ate the fish they caught, it was assumed that 20% of the fish caught (and eaten) from within the AOC were caught in Peninsula Harbour.

The screening level HHRA concluded that methylmercury in fish caught in the AOC is not likely to pose an unacceptable health risk to adult anglers or their household members (i.e., toddlers, children and adolescents). However, PCBs in fish caught in the AOC are predicted to pose an unacceptable health risk to adult anglers and their household members. The most effective way to mitigate this potential risk would be to improve awareness of the fish consumption advisory among of the residents of Marathon and nearby communities. In general, consumption of any fish caught from the AOC should be avoided.

Sediment management goals were developed based on four types of management criteria: 1) risk-based management goals; 2) guideline-based management goals; 3) background-based management goals; and 4) source control of hot spots. The first set, risk-based sediment management goals, was based on the outcome of foregoing risk assessment. In particular, those ecological and human receptors and scenarios found to drive risks were used to back-calculate fish tissue concentrations that would not be expected to cause adverse effects in any receptors (i.e., risk-based target tissue concentrations). A food web bioaccumulation model based on the work of Grapentine et al. (2005) and Hope (2003) was then used to estimate sediment concentrations that would be required to attain those risk-based target fish tissue concentrations (i.e., sediment management goals). The second set of goals, guideline-based management goals, was similarly estimated using the food web model, although in this case, the model predicted the sediment concentration that would be required to attain TRGs. The third set of goals, background-based management goals, is the set of modeled sediment concentrations that would be required to attain fish tissue concentrations of methylmercury and PCBs consistent with local background levels (i.e., 0.18 mg/kg mercury and 0.40 mg/kg PCBs). The fourth set of goals is based on hot spots of total mercury in sediment, defined by fixed concentrations of total mercury in sediment.

In order to characterize the area and volume of sediment potentially warranting management under the four sets of sediment management goals developed above, the spatial distribution of mercury and PCBs in JC and RPH was first considered under current conditions. Simulations of various remedial scenarios were then employed to determine the area and volume of sediment warranting remediation in order to attain the sediment management goals in JC and RPH. Based on the available data on the distribution of methylmercury in the AOC, specific management areas were defined that would achieve sediment management goals based on predicted ecological risks, human health risks, guidelines, background, and hot spots. Volumes associated with those areas were then estimated based on the available information on the depth of sediment in the AOC. Extensive areas and volumes of sediment would warrant management in order to mitigate risks or achieve either TRGs or background levels in fish tissue.

Although hot spot remediation scenarios are based on source control, rather than risk mitigation, management of sediment containing elevated concentrations of mercury and total PCBs is expected to reduce risks. In particular, hot spot remediation is predicted to reduce HQs for fish from methylmercury by 7% to 62%, depending on the mercury concentration used to define the hot spot and the fraction of time that fish are assumed to forage in JC and RPH. If the hot spot is defined by total mercury in sediment greater than either 2 mg/kg or 3 mg/kg, residual HQs for fish are not predicted to exceed the target HQ of 1, regardless of the amount of time that fish are assumed to forage in JC and RPH. Hot spot remediation is predicted to reduce HQs for mink from total PCBs by 4% to 44%, depending on the mercury concentration used to define the hot spot and the fraction of time that mink are assumed to forage in JC and RPH. Mink HQs do not exceed 1 when the hot spot is defined by 2 mg/kg or 3 mg/kg, regardless of the proportions of prey derived from JC and RPH. In summary, if the hot spot is defined by the total mercury concentration of 3 mg/kg in sediment, risks are expected to be reduced to acceptable levels for fish and mink. Remediation of sediment containing concentrations of total mercury equal to or greater than 3 mg/kg would cut the concentration of methylmercury in JC in half, from 0.0052 mg/kg to 0.0027 mg/kg.

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Data Used in Risk Assessment

Appendix C:
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Appendix D:

Data Summary Report:
Evaluation of Mink and River Otter Habitat Suitability in the Peninsula Harbour Area of Concern

Appendix E:

Appendix F:

Fish Consumption Survey

Marathon, ON and Pic River Reserve