Human Health and Ecological Risk Assessment of the PCB Area at the Upper Trinity South (New Harbour) Waste Disposal Site, New Harbour Barrens, Newfoundland and Labrador

FINAL REPORT

July 2013

Project No: 13-7515-1000

Submitted to:
Government of Newfoundland and Labrador, Department of Environment and Conservation

Submitted by:
Dillon Consulting Limited

DILLON
CONSULTING

66 Kenmount Road
Suite 203
St. John's
Newfoundland
Canada
A1B 3V7
Telephone
(709) 754-2374
Fax
(709) 754-2380

446 Newfoundland Drive
St. John's
Newfoundland
Canada
A1A 4G7
Telephone
(709) 739-7890
Fax
(709) 739-7601

July 24, 2013

Government of Newfoundland and Labrador
Department of Environment and Conservation
4th Floor Confederation Building, West Block
P. O. Box 8700
St. John's, NL A1B 4J6

ATTENTION: Jennifer Strickland, B.Sc.
Environmental Scientist

Human Health and Ecological Risk Assessment of the PCB Area at the Upper Trinity South (New Harbour) Waste Disposal Site, New Harbour Barrens, NL

We are pleased to provide a copy of the final Human Health and Ecological Risk Assessment report for the polychlorinated biphenyl area at the Upper Trinity South Waste Disposal Site at New Harbour Barrens, Newfoundland.

Should you have any questions or concerns upon your review of this document, please do not hesitate to contact either of the undersigned.

Yours truly,

DILLON CONSULTING LIMITED

Randy Fancey

Randy Fancey, CET
Project Manager

RF/RW:rbc
Our File: 13-7515-1000

Y:\PROJECTS\FINAL\137515 New Harbour HHERA\Reports\Final Report\New Harbour HHHERA Report (Final).docx

Glossary of Terms

ATSDR	Agency of Toxic Substances and Disease Registry
ADI	Acceptable Daily Intake
AUF	Area Use Factor
BW	Body Weight
CALA	Canadian Association for Laboratory Accreditation
CalEPA	California Environmental Protection Agency
CEQG	Canadian Environmental Quality Guidelines
CSM	Conceptual Site Model
CCME	Canadian Council of Ministers of the Environment
COPC	Chemical of Potential Concern
CSF	Cancer Slope Factor
CSQG	Canadian Soil Quality Guidelines
DQRA	Detailed Quantitative Risk Assessment
EC	Environment Canada
ED	Exposure Duration
EF	Exposure Frequency
EcoSSL	Ecological Soil Screening Level
EHQ	Ecological Hazard Quotient
EPA	Environmental Protection Agency
EPC	Exposure Point Concentration
ER	Exposure Rate
ERA	Ecological Risk Assessment
ESA	Environmental Site Assessment
FAL	Freshwater Aquatic Life
FCSAP	Federal Contaminated Sites Action Plan
FD	Field Duplicate
GW	Groundwater
HHERA	Human Health and Ecological Risk Assessment
HHRA	Human Health Risk Assessment
HQ	Hazard Quotient
IACR	Index of Additive Cancer Risk
IARC	International Agency for Research on Cancer
ILCR	Incremental Lifetime Cancer Risk
ISQG	Interim Sediment Quality Guideline
LADD	Lifetime Average Daily Dose
LD	Laboratory Duplicate
LLPDE	Linear Low Density Polyethylene
LOAEL	Lowest Observed Adverse Effect Level
LOE	Line of Evidence
MRL	Minimal Risk Level
NL	Newfoundland and Labrador
GNLDEC	Newfoundland and Labrador Department of Environment and Conservation
NOAEL	No Observed Adverse Effect Level
OMOE	Ontario Ministry of the Environment

PAH	Polycyclic Aromatic Hydrocarbon
PCB	Polychlorinated biphenyl
PDI	Permissible Daily Intake
PEF	Potency Equivalence Factor
PEL	Probable Effects Level
PM	Particulate Matter
PPE	Personal Protective Equipment
PQRA	Preliminary Quantitative Risk Assessment
PWGSC	Public Works and Government Services Canada
OEHHA	Office of Environmental Health Hazard Assessment
QA/QC	Quality Assurance / Quality Control
RAF	Relative Absorption Factor
RBA	Relative Bioavailability
RBCA	Risk Based Corrective Action
RBSLs	Risk Based Screening Levels
RDL	Reported Detection Limit
RfD	Reference Dose
RMP	Risk Management Plan
RPD	Relative Percent Difference
ROC	Receptor of Concern
RQ	Risk Quotient
SAR	Species At Risk
SARA	*Species at Risk Act*
SCC	Standards Council of Canada
Sd	Standard Deviation
S-OA	Soil to Outdoor Air
SoQG	Soil Quality Guideline
SSTL	Site-Specific Target Level
TCA	Tolerable Concentration in Air
TDI	Tolerable Daily Intake
THQ	Target Hazard Quotient
TPH	Total Petroleum Hydrocarbons
TSP	Total Suspended Particulates
TRV	Toxicity Reference Value
UCL	Upper Confidence Limit
UF	Uncertainty Factor
USEPA	United States Environmental Protection Agency
VOC	Volatile Organic Compound
WDS	Waste Disposal Site
WHO	World Health Organization
WOE	Weight Of Evidence

EXECUTIVE SUMMARY

Dillon Consulting Limited (Dillon) was retained by the Government of Newfoundland and Labrador Department of Environment and Conservation (GNLDEC) to conduct a Human Health and Ecological Risk Assessment (HHERA) associated with residual Polychlorinated Biphenyl (PCB) impacted soil within a localized area (referred to as the "PCB Area") of the Upper Trinity South (New Harbour) Waste Disposal Site (WDS), New Harbour Barrens, Newfoundland and Labrador.

The objective of the HHERA was to determine whether or not the residual PCB concentrations in soil that remain at the WDS pose an unacceptable human health or ecological risk. Two scenarios were conducted for the HHERA: the Current Scenario (As Is - Outdoor Site Visitor (Trespasser)) and the Future Scenario (Post-Capping).

The WDS operated from the early 1970s (exact date is unknown) up until November 2009, accepting waste from residents and businesses from the surrounding communities. A quantity of PCB impacted scrap metal and transformer casings were disposed of at the WDS between 1992 and 1995. The PCB impacted metal and transformer casings were buried in the northwest portion of the WDS, which has been commonly referred to as the PCB Area.

Several environmental site assessment (ESA) and monitoring programs have been conducted since 1996 by various consulting firms. These programs have focussed on delineating and assessing PCBs in soil, groundwater, surface water and sediment, as well as, assessing PCB concentrations in fish down-gradient of the WDS. Two remediation programs were also conducted whereby PCB impacted soil was excavated and disposed off-site at provincially approved disposal facilities. However, PCBs remain in soil at concentrations that exceed the Canadian Council of Ministers of the Environment, Canadian Soil Quality Guideline concentrations.

The HHERA was conducted based on information and analytical data collected from the previous ESA, remediation and monitoring programs. Dillon also collected one surficial soil sample to delineate the PCB impacts in an area not previously sampled, due to the presence of boulders.

The following presents a brief summary of the HHERA outcomes for the two evaluated scenarios:

HHRA Conclusions

Current Scenario (As Is - Outdoor Site Visitor (Trespasser))

- The human health risk assessment (HHRA) indicated the overall potential for human health risk is considered to be low.
- The current PCB concentrations in soil do not pose an immediate human health concern.

Future Scenario (Post-Capping)

- The HHRA for this scenario indicates there will be no unacceptable human health risks once the engineered cap is in place.

ERA Conclusions

Current Scenario (As Is - Outdoor Site Visitor (Trespasser))

- The ecological risk assessment (ERA) indicated the potential for ecological risks is considered to be low.
- Based on the ERA conclusions, there is no apparent need for mitigation of the PCB Area with respect to potential ecological risks.

Future Scenario (Post-Capping)

- The ERA for this scenario indicates there will be a negligible potential for ecological risks once the engineered cap is in place.

To date, PCB concentrations have not been detected in the off-site surface water, sediment or fish samples. As such, there is no potential off-site human health or ecological risk associated with the PCBs in soil at the WDS.

Dillon recommends that an engineered cap be installed over the PCB Area of the WDS. It is understood that the GNLDEC are planning to install an engineered cap over the PCB Area in the Fall of 2013.

TABLE OF CONTENTS

LIST OF TABLES, FIGURES, AND APPENDICES

TABLES

FIGURES

APPENDICES

1.0 INTRODUCTION

Dillon Consulting Limited (Dillon) was retained by the Government of Newfoundland and Labrador Department of Environment and Conservation (GNLDEC) to conduct a Human Health and Ecological Risk Assessment (HHERA) associated with residual Polychlorinated Biphenyl (PCB) impacted soil within a localized area (referred to as the "PCB Area") of the Upper Trinity South (New Harbour) Waste Disposal Site (WDS), also referred to as the "Site", New Harbour Barrens, Newfoundland and Labrador (NL) (Figures 1 and 2, Appendix A).

A number of environmental site assessments (ESAs), remediation and monitoring programs have been conducted at the WDS, including the PCB Area, since 1996. While PCB impacted soil was removed from the PCB Area in 2008 and 2010, residual PCBs remain in soil at concentrations that exceed the Canadian Council of Ministers of the Environment (CCME), Canadian Soil Quality Guideline (CSQG) concentration for a commercial site.

As part of the overall and on-going WDS closure plan/program, the GNLDEC is interested in determining if the residual PCB concentrations in soil pose a potential unacceptable human health and/or ecological risk. At the request of the GNLDEC, the HHERA was conducted for two scenarios, which were identified as follows:

1. Current Scenario (As Is - Outdoor Site Visitor (Trespasser)); and
2. Future Scenario (Post-Capping).

The following report presents the objectives and scope of the HHERA (Section 2.0), site description and history (Section 3.0), previous environmental programs (Section 4.0), data gap analysis (Section 5.0), soil sampling (Section 6.0), PCB distribution (Section 7.0), human health risk assessment (Section 8.0), ecological risk assessment (Section 9.0), conclusions (Section 10.0) and recommendations (Section 11.0). Supporting documents are presented in Appendices A to F.

2.0 OBJECTIVES AND SCOPE OF THE HHERA

The objectives of the HHERA are to:

- Review available documentation provided by the GNLDEC, conduct a data gap analysis and provide recommendations to address identified data gaps to support the HHERA, if warranted;
- Determine if there are potential unacceptable human health and ecological risks posed by the presence of residual PCBs in soil at the PCB Area;
- Determine if PCBs have migrated off-site into terrestrial and aquatic media, and if so, evaluate potential unacceptable off-site human health and ecological risks;

- Based on the outcomes of the HHERA, determine if further action(s) is necessary to mitigate potential human health and/or ecological risks, and if so, develop a Risk Management Plan (RMP) for the Site; and,
- Prepare a report summarizing the approach, methodology, findings, outcomes and recommendations of the HHERA.

To achieve these objectives, the HHERA was conducted at a combined screening level and preliminary quantitative level of effort in accordance with current Health Canada, Environment Canada and CCME guidance documentation.

The HHERA is limited to the PCB Area of the WDS and is also limited to the evaluation of PCBs only. No other areas, contaminants or environmental issues associated with the WDS are considered or evaluated herein.

The HHERA was conducted using existing on-site and off-site analytical PCB data (i.e., soil, groundwater, surface water, sediment and fish tissue) collected by others[1]. A sampling and analytical program or supplementary ESA was not within the scope of work for this HHERA. Based on the previous assessment, remediation and monitoring programs, available analytical data was considered to be appropriate, in terms of both quantity and quality, to conduct a HHERA. The PCB Area, including the up-gradient and down-gradient sample locations (for determining if PCBs have migrated off-site), was considered by Dillon to have been well characterized.

The HHERA evaluates current and potential future exposures associated with the residual PCB concentrations in soil at the PCB Area. It does not evaluate potential exposures and risks that may have been present in the past, which is a common practice of any risk assessment. Typically, both human health and ecological risk assessments are prospective in nature and focus on current and future conditions. Human health or ecological risk assessments of contaminated sites are rarely conducted in a retrospective manner. This reflects the fact that reliable data for historical exposures is typically lacking for most sites. Where such historical data does exist, it is often limited (in both quantity and quality) and can frequently be unreliable due to reporting limitations, sampling and analytical deficiencies or uncertainties.

3.0 SITE DESCRIPTION AND HISTORY

The WDS is located on Route 73 on the Avalon Peninsula of Newfoundland, approximately six kilometres (km) northeast of New Harbour, NL (Figures 1 and 2, Appendix A).

[1] With the exception of one soil sample collected by Dillon in March 2013 for PCB Area delineation purposes.

The WDS operated from the early 1970s (exact date is unknown) up until November 2009, accepting waste from residents and businesses from the surrounding communities of Blaketown, Dildo, Green's Harbour, Hopewell, Markland, New Harbour, Old Shope and South Dildo.

The WDS was operated by a local contractor, who typically placed the waste in open cells or pits. It is also understood that open burning was conducted at the WDS to reduce garbage volume and control pests. In addition to accepting municipal waste, the WDS also had a designated area for the disposal of bulk metals (including car wrecks) and an area used by a local seal processing plant for the disposal of fat, seal pelt trimmings, sawdust and sludge. A quantity of PCB impacted scrap metal and transformer casings, which originated from the Makinsons scrap yard, was also disposed of at the WDS between 1992 and 1995.

The PCB impacted metal and transformer casings were buried in the northwest portion of the WDS, which has been commonly referred to as the PCB Area (Figure 3, Appendix A). Based on previous reports, the PCB Area covers an area of approximately 913 m^2. There are generally steep embankments sloping down from the plateau to the original ground surface along the outer perimeter of the WDS. Vegetation (i.e., grass) is sparse and patchy over the PCB Area. Photographs of the WDS and the PCB Area are presented in Appendix B.

The WDS is unlined. Interceptor ditches and a leachate collection pond were installed between 2006 and 2007 to manage potential leachate impacts. In addition to the leachate collection and retention system, seven groundwater monitoring wells were installed around the WDS to assess and monitor potential leachate impacts in groundwater.

There were no permanent buildings at the WDS during active operations. As such, there was no on-site potable water supply (i.e., dug or drilled well).

The WDS was closed in November 2009 with fencing and concrete barricades installed at that time to prevent the future disposal of waste at the WDS.

A soil cover, using clean imported cover material, was recently placed on the waste material in 2012 and 2013 to facilitate settling. Maintenance of this cover was also completed in 2012-2013. It is understood that the GNLDEC intends to install a final engineered cap over the PCB Area, which will include a linear low density polyethylene (LLDPE) liner and clean imported cover materials

3.1 Current and Future Land Use

The WDS is classified as a commercial property based on past activities. It is assumed the WDS will remain commercial, even after the closure activities have been completed.

Although occasional transient access by trespassers/visitors is possible, the majority of on-site activities will likely be limited to on-going and future installation and maintenance of the engineered cap, as well as, the leachate control system and environmental monitoring.

3.2 Surrounding Area

Route 73 borders the WDS to the north and provides direct access into the Site. Undisturbed areas, consisting of typical wooded and barren areas surround the Site to the east, south and west (Figure 3, Appendix A). Photographs of the areas surrounding the WDS are presented in Appendix B. Vegetation in these areas appears healthy, recognizing that site visits, as part of this project, were conducted in March and April 2013. There are also several ponds in the vicinity of the Site, an unnamed pond (220 metres (m) southeast of the Site), Three Corner Pond (350 m south of the Site), Loo Pond (550 m southwest of the Site), as well as, a small seasonal surface water body located approximately 50 m west of the Site.

The closest residential dwellings are located around Denny's Pond, approximately two km west of the WDS. There are also several seasonal cabins located around Gull Pond, approximately 250 m north northeast of the Site, across Route 73. The location of the residential dwellings and seasonal cabins are presented in Figure 2, Appendix A.

3.3 Topography and Surficial/Bedrock Geology

Information on topography and surficial/bedrock geology was obtained from previous reports, topographical/geological maps, aerial photographs and the two site visits conducted by the Dillon representative in March and April 2013.

The topography in the vicinity of the WDS generally slopes downward to the southwest. The terrain is lightly wooded to the east and west and heavily wooded to the south.

The surface geology in the vicinity of the WDS is mapped as concealed bedrock. Bedrock is mainly concealed by vegetation; patches of till, sand and gravel or bog (usually less than 1.5 m thick), and exposed bedrock is common, but makes up less than 50% of the ground surface. Bedrock is mapped as fluviatile and shallow marine siliciclastic sedimentary rocks, including minor unseparated limestone and bimodal volcanic rocks (Signal Hill Group; parts of Musgravetown, Long Harbour, Connaige Bay, Marystown and Love Cove groups).

Based on the monitoring well logs for MW-01 to MW-07 (Implementation of the Leachate Control System, AMEC, March 2007), the surficial soil was logged as 0.2 to 0.6 m (with an average depth of 0.4 m) of topsoil/rootmat consisting of sand and gravel, with traces to some fines and cobbles and boulders. This unit is underlain by a glacial till consisting of grey sand and gravel with traces of cobbles and

boulders. The glacial till extended to depths ranging from 0.9 to 2.1 m below surface (with an average depth of 1.6 m below surface). Bedrock was logged as highly fractured grey sandstone.

Based on the test pit logs for NH-TP-01 to NH-TP-16 (AMEC, 2013) soil conditions within the PCB Area generally consists of approximately 3 to 4 m of brown/grey silty sand and gravel with cobbles and boulders (considered to be coarse grained). Various depths of fill/glacial till were logged. Various types of debris and waste materials such as sawdust, metal, tires, transformer casings, seal pelts, and creosote timbers were also observed within the PCB Area (in addition to transformer casings). Organic peaty soil has also been observed in many locations within and surrounding the WDS.

3.4 Groundwater Use and Hydrology

There is no water supply well(s) or potable groundwater use within the WDS. Therefore, groundwater at the WDS is considered to be non-potable.

Potable water for the residential dwellings around Denny's Pond is supplied by private water supply wells (i.e., drilled wells). Although unconfirmed, a local well driller indicated potable water for the cabins around Gull Pond is obtained from dug wells. It is noted that Gull Pond is located up-gradient of the WDS and is unlikely to be affected by the WDS.

Groundwater flow direction at the Site is toward the south, with an approximate hydraulic gradient of 0.038, as presented in the 2011-2012 Annual Report (SNC Lavalin, 2012). Depth to groundwater, as measured in the seven monitoring wells around the WDS, typically ranges between 0.9 to 1.6 m, with the exception of MW-02, which typically measures between 3.0 to 3.9 m.

Surface water flow direction in the vicinity of the WDS is to the southwest and then west as surface water from Three Corner Pond flows towards and into Loo Pond (Figure 1, Appendix A). Surface water from Loo Pond ultimately flows into Denny's Pond, which is located approximately two (2) km west of the WDS. Denny's Pond also receives surface water through a stream that flows under Route 73 (west of the WDS) from Gull Pond, which is located up-gradient of the WDS.

4.0 PREVIOUS ENVIRONMENTAL PROGRAMS

Several ESA, remediation and monitoring programs have been conducted at the WDS (including the PCB Area) since 1996, which have included: test pitting; installing groundwater monitor wells; surveying; soil, groundwater, surface water and fish sampling; etc. The following presents a brief bullet format summary of the PCB specific information from these programs:

1996 (Harris and Associates Limited)

- Although groundwater and surface water samples were collected, the samples were not analyzed for PCBs.
- This report was not available, but the information was presented in the report Environmental Testing, Final Report, SGE Acres Limited, February 2003.

2002 (Department of Environment)

- PCBs were not detected in the surface water samples identified as the Steady, Three Corner Pond and Denny's Pond.
- This report was not available, but the information was presented in the report Environmental Testing, Final Report, SGE Acres Limited, February 2003.

Environmental Testing, Final Report, February 2003 (SGE Acres Limited)

- Six test pits (TP #1 to TP #6) were advanced around the WDS.
- Six soil samples were analyzed for PCB analysis. PCB concentrations were not detected (i.e., <0.05 mg/kg) in these samples.
- Two sediment samples (Sed A and Sed B) were collected and analysed for PCBs. PCB concentrations were not detected (i.e., <0.05 mg/kg) in these sediment samples.
- Two surface water samples (Sed A and Sed B) were collected and analysed for PCBs. PCB concentrations were not detected (i.e., <0.05 ug/l) in these surface water samples.

Recommendations: It was recommended that the spatial extent of contamination coming from the dump be determined and assess whether or not the dump is affecting fish and fish habitat. It was also recommended to assess whether or not the dump is having an impact on the health of people living in the area.

Part I, Phase II Environmental Testing, May 2003 (SGE Acres Limited)

- SGE Acres excavated a trench in an area of the dumpsite where old transformer casings were buried.
- Six soil samples (#1 to #6) were collected from the trench and were submitted to Environmental Services Laboratory Incorporated in Sydney, Nova Scotia for PCB analysis. A PCB concentration of 52 mg/kg in sample #3 exceeded the 2004 CCME Canadian Environmental Quality Guideline (CEQG) concentration (33 mg/kg) for an industrial site and the Canadian Environmental Protection Act Chlorobiphenyls Regulation concentration (50 mg/kg). PCB concentrations were not detected (<1 mg/kg) in the other five analyzed soil samples.

Recommendations: It was recommended that PCB impacted soil with concentrations greater than 50 mg/kg be excavated and disposed off-site at a licensed disposal facility.

Part II, Phase II Environmental Sampling, March 2004 (SGE Acres Limited)

- Although surface water samples were collected from the down-gradient ponds, including an up-gradient background location, the samples were only sampled for metals.

Design of Leachate Control System, June 2006 (AMEC Earth and Environment)

- Five test pits (TP1 to TP5) were advanced in the vicinity of the PCB impacted soil area.
- Fifteen composite soil samples were submitted to the AMEC Earth & Environmental Analytical Laboratory for PCB analysis. PCB concentrations were detected in all fifteen analyzed soil samples at concentrations below the applicable 2004 CCME CSQG (Commercial) concentration (33 mg/kg).
- These results did not duplicate the findings reported during the previous SGE Acres soil sampling program for one soil sample (#3) which exceeded the applicable 2004 CCME CEQG concentration for PCBs.

Recommendations: It was recommended that water samples be collected from the surface, mid-column and bottom layers in conjunction with sediment samples from the deepest areas of Denny's pond and Three Corner Pond. A habitat survey was also recommended for the Gully Stream, the Steady and the Three Corner Pond and any interconnecting streams to classify the quality and quantity of available salmonoid habitat. Lastly, it was also recommended that fish samples be collected and analyzed.

Design of Leachate Control System Revisions, March 2007(AMEC Earth and Environment)

- This report provides updated recommendations for future closure activities and capping based on leachate control measures that had been implemented. No assessment or sampling activities were conducted specifically relating to PCBs.

Implementation of the Leachate Control System, March 2007 (AMEC Earth and Environmental Limited)

- Seven groundwater monitoring wells (MW-01 to MW-07) were installed in November and December 2006. PCB concentrations were not detected in the two analyzed groundwater samples (MW-03 and MW-05).
- Two test pits (TP6 and TP7) were advanced in November 2006.
- Ten soil samples were collected and analyzed for PCBs. One soil sample (TP6 SA-4) had a PCB concentration of 66.7 mg/kg which exceeded the CCME CSQG (Commercial) concentration of 33 mg/kg. PCBs were detected in the other nine analyzed samples at concentrations ranging from 0.052 to 30.1 mg/kg, all of which are below the CCME CSQG (Commercial) concentration.

Recommendations: No recommendations were presented.

2006 Fish Sampling Program, May 2007 (AMEC Earth and Environment)

- Ten control samples from Gull Pond (up-gradient of the WDS) and ten samples from down-gradient of the WDS were collected and analyzed for PCBs. PCB concentrations were not detected in the control or down-gradient fish samples and were therefore below the Canadian Food Inspection Agency (CFIA) and the U.S. Food and Drug Administration Level concentrations for human consumption.

Recommendations: No recommendations were presented.

2007 Groundwater and Surface Water Sampling Program, March 2008 (AMEC Earth and Environmental)

- Groundwater samples were collected from the seven monitoring wells (MW-01 to MW-07) in November 2007. PCB concentrations were not detected (<0.04 ug/L) in these groundwater samples.
- Two surface water samples (the "Pond" and the "Stream") were collected in November 2007. PCB concentrations were not detected (<0.04 ug/L) in these surface water samples.

Recommendations: It was recommended to continue to monitor groundwater and surface water at the Site to evaluate the effectiveness of the leachate control system.

2008 Groundwater and Surface Water Sampling Program, March 2009 (AMEC Earth and Environmental)

- Groundwater samples were collected from the seven monitoring wells (MW-01 to MW-07) in May 2008. PCB concentrations were not detected (<0.04 ug/L) in these groundwater samples.
- Two surface water samples (the "Pond" and the "Stream") were collected in May 2008. PCB concentrations were not detected (<0.04 ug/L) in these surface water samples.

Recommendations: It was recommended to continue to monitor groundwater and surface water at the Site to evaluate the effectiveness of the leachate control system.

2008-2009 Annual Report of Activities, AMEC, March 2009 (AMEC Earth and Environmental)

- **Phase I Soil Removal Program (September 2008):**

  Location A:

  - The excavated area with PCB impacted soil was approximately 5.0 x 5.0 m and extended to an average depth of 3.0 m.
  - The top 2.0 m of soil was excavated and stockpiled on-site.
  - The soil from 2.0-3.0 m (approximately 20.3 tonnes) was excavated and loaded directly into tandem trucks, and was then transported and disposed off-site at the Universal Environmental Services Inc. (UESI) facility in Sunnyside, NL.

- Bedrock was encountered in the south end of the excavation. Groundwater was not encountered in the excavation.
- Five soil samples (SA-1 to SA-5) were collected from the final excavation limits and were submitted to Maxxam for PCB analysis. The PCB concentrations in these samples ranged from 55 to 220 mg/kg, all of which exceeded the CCME CSQG (Commercial) concentration of 33 mg/kg.
- A soil sample was also collected from the stockpiled soil (Stockpile 1) and was submitted to Maxxam for PCB analysis. The PCB concentration in this sample was 150 mg/kg, which exceeded the CCME CSQG (Commercial) concentration of 33 mg/kg.

Location B:

- The excavated area with PCB impacted soil was approximately 5.0 x 5.0 m and extended to an average depth of 3.0 m.
- The top 2.0 m of soil was excavated and stockpiled on-site.
- The soil from 2.0-3.0 m (approximately 23.27 tonnes) was excavated and loaded directly into tandem trucks, which was then transported and disposed off-site at the UESI facility in Sunnyside, NL.
- Bedrock and groundwater were encountered within this excavation.
- Five soil samples (SA-6 to SA-10) were collected from the final excavation limits and were submitted to Maxxam for PCB analysis. The PCB concentrations in these samples ranged from 0.64 to 15 mg/kg, all of which are below the CCME CSQG (Commercial) concentration of 33 mg/kg.
- A soil sample was also collected from the stockpiled soil (Stockpile 2) and was submitted to Maxxam for PCB analysis. The PCB concentration in this sample was 0.83 mg/kg, which was below the CCME CSQG (Commercial) concentration of 33 mg/kg.

In total, approximately 43.57 tonnes of PCB impacted soil was excavated from Locations A and B, transported and disposed off-site at the UESI facility in Sunnyside, NL.

- **Phase II Soil Removal Program (October 2009)**

  Location A

  - The Phase I excavation was extended approximately 3.5 m to the north, east and west. The excavation was extended to an average depth of approximately 3.0 m. The expanded excavation was approximately 12.0 m x 8.5 m.
  - An additional 76.78 tonnes of PCB impacted soil was excavated, transported and disposed off-site at the UESI facility in Sunnyside, NL.
  - Bedrock and groundwater were not encountered within the excavation.
  - Ten additional soil samples (SA-11 to SA-20) were collected from the final excavation limits and were submitted to Maxxam for PCB analysis. PCB concentrations in five of the ten samples

ranged from 58 to 600 mg/kg, which exceeded the CCME CSQG (Commercial) concentration of 33 mg/kg.

- Two soil samples were collected from the surrounding overburden and were submitted to Maxxam for PCB analysis. The PCB concentrations in these samples were 10 and 6.6 mg/kg, which are below the CCME CSQG (Commercial) concentration of 33 mg/kg. The overburden was subsequently used as backfill material.
- One addition soil sample was also collected from the Phase I, Location A stockpiled material. The PCB concentration for this sample was 110 mg/kg, which exceeded the CCME CSQG (Commercial) concentration of 33 mg/kg.
- At the request of the GNLDEC, Location A was backfilled with PCB impacted soil, including soil that was initially excavated and stockpiled from Location A during the Phase I program. Approximately 100 tonnes of PCB contaminated soil were placed in a cell lined with 6 mil plastic. The contaminated soil was from the original stockpiled material. In addition to the plastic cover, orientated strand board was also placed over the top of the contaminated soil and plastic to mark the boundary of the contaminated soil. Approximately 150 tonnes of overburden was then placed over the contaminated soil.
- A plastic (polyethylene) cover, which was secured with a tarp and boulders, was placed over the entire backfilled area.

Location B

- Location B was backfilled with the stockpiled material (from 0-2 m) and surrounding overburden.

Supplemental Soil Sampling (January 2009)

- Five trenches were excavated and extended out from the previously excavated Location A.
- In total, forty-four soil samples were collected from the trenches and were submitted to Maxxam for PCB analysis. PCB concentrations in three of these samples (Trench 2: SA-1, Trench 3: SA-5 and Dup-3) exceeded the CCME CSQG concentration of 33 mg/kg.
- Numerous transformer casings and scrap metal were observed in some of the trenches.

Recommendations:

- Continue to monitor groundwater and surface water quality at the Site for PCBs, specifically at MW-01 to MW-06, Pond, Stream and the Ditch.
- Collect sediment samples from the leachate collection pond, interception ditches and the down-gradient stream for PCB analyses;
- Conduct a test pitting program along the perimeter of the landfill, in the vicinity of the existing monitoring wells, to assess the levels of PCBs in surface and subsurface soil at that area of the Site;
- Conduct additional soil assessment in the area of the Location A to further delineate the extent of PCB impacts in that area;

- Conduct a Human Health Preliminary Quantitative Risk Assessment (PQRA) and Screening Level Ecological Risk Assessment (SLERA) to determine whether or not the PCB concentrations in soil pose a potential risk to human and ecological receptors;
- Following the risk assessment, develop a Remedial Action Plan/Risk Management Plan (RAP/RMP) for the Site;
- Obtain the services of a survey contractor to survey the "top of casing" and "ground surface" elevations for the existing monitoring wells to further evaluate the direction of groundwater flow throughout the Site; and
- Repair or re-install monitoring well MW-05.

Removal of PCB – Impacted Material, January 2011 (AMEC Earth & Environmental)

- AMEC prepared an Invitation to Tender (ITT or Tender Documents) for the excavation, transportation and off-site disposal of approximately 120 tonnes of additional PCB impacted soil. The PCB impacted soil to be removed consisted of the impacted soil that was initially excavated and stockpiled from Location A during the Phase I and later used as backfill material in Phase II.
- In total, 136 tonnes of PCB impacted soil within the plastic cell was excavated, transported and disposed off-site at the Horizon Environmental Inc. facility in Grandes-Piles, Quebec.
- Confirmatory soil samples were not collected as part of this program.
- The excavated area was backfilled with clean imported fill. A sample of the imported fill was collected and analyzed for PCBs. A PCB concentration was not detected (<0.05 mg/kg) in the imported fill sample.

Closure Plan, March 2011 (AMEC Earth & Environmental)

- Although no specific activities were conducted to further assess PCBs at the Site, the following recommendations were presented:
    - Advance additional test pits to delineate the boundaries of the transformer disposal area; and
    - Conduct a Human Health and Ecological Risk Assessment to assess the potential risks associated with exposure to PCBs in environmental media to human health and ecological receptors.

2010-2011 Annual Report of Activities, March 2011 (AMEC Earth & Environmental)

- Groundwater samples were collected from seven existing monitoring wells (MW-01 and MW-03 to MW-08) in December 2010 and were submitted to Maxxam for PCB analysis. PCB concentrations were not detected (<0.05 mg/L) in the analyzed groundwater samples.
- Two surface water samples were collected in December 2010 from the SW-POND and STREAM and were submitted to Maxxam for PCB analysis. PCB concentrations were not detected (< 0.05 ug/L) in the surface water samples.
- Three sediment samples, including two (2) background samples, were collected in December 2010 from the POND-SED, STREAM-SED, DITCH-SED and were submitted to Maxxam for PCB

analysis. A PCB concentration of 2.8 mg/kg was detected in the sample DITCH-SED, which exceeds the CCME CSedQG (Interim Sediment Quality Guidelines (ISQG) and the Probable Effects Level (PEL)) concentrations. PCB concentrations were not detected (< 0.05 mg/kg) in the other two sediment samples.

2011-2012 Annual Report of Activities, July 2012 (SNC-Lavalin)

- Groundwater samples were collected in December 2011 from the eight existing monitoring wells (MW-01 to MW-08) and were submitted to Maxxam for PCB analysis. PCB concentrations were not detected (<0.05 mg/L) in the nine analysed groundwater samples (including a duplicate sample analysis).
- Two surface water samples were collected in December 2011 from the Pond and the Stream and were submitted to Maxxam for PCB analysis. PCB concentrations were not detected in the two surface water samples.

Recommendations: Although not specifically identified as a recommendation, a post decommissioning monitoring and maintenance plan was identified, which included continued groundwater, surface water and sediment monitoring.

Test Pitting and Soil Sampling Program, February 2013 (AMEC Environment and Infrastructure)

- Sixteen test pits (NH-TP-01 to NH-TP-16) were advanced in February 2013 to better delineate the PCB impacted area.
- In total, thirty-four soil samples were collected and analyzed for PCBs (two samples per test pit). PCB concentrations in the analyzed soil samples from NH-TP-01-SS2 and NH-TP-07-SS2 exceeded the CCME CSQG concentration. The PCB concentrations in the other analyzed samples were either non-detect or the concentrations were below the CCME CSQG concentration.
- Transformer casings were observed in test pits NH-TP-01, NH-TP-05 and NH-TP-07.

5.0 DATA GAP ANALYSIS

As previously indicated the HHERA was conducted using information and analytical data from the previous assessment, remediation and monitoring reports. However, based on a review of the available documents, Dillon identified one data gap. The PCB impacted area was not delineated along the northwest portion of the PCB Area, near the interceptor ditch. Reportedly, this area had not been previously sampled because of the presence of large boulders.

Dillon recommended a surficial soil sample be collected from this area to delineate the PCB impacted area, which could also be used to demonstrate PCBs are not migrating off-site and impacting the down-gradient environment which would be beneficial to support the HHERA outcomes.

The GNLDEC subsequently authorized Dillon to collect a surficial soil sample from this location.

6.0 SOIL SAMPLING

6.1 Soil Sampling Methodology

Dillon conducted a site visit on April 2, 2013 and collected a single soil sample (SS1A (0.3-0.8 m)) to delineate the north-eastern most portion of the PCB Area. The soil sample location is presented in Figure 4, Appendix A.

A shallow test hole (STH) was manually advanced using a shovel and trowel to a depth of approximately 0.8 m below surface. It is noted this STH was advanced on undisturbed original surface near the toe of the embankment of the WDS. Although the soil sample was collected from 0.3-0.8 m below surface, this depth is equivalent to approximately 2.7-3.2 m below surface in the WDS (i.e., a slightly lower elevation than the impacted PCB soil at approximately 2-3 m below surface within the WDS).

The shovel and trowel were cleaned, using diluted Simple Green™ and then rinsed with bottled water, before advancing the test hole then again before collecting the sample. The soil sample was placed directly into a clean laboratory supplied sample bottle, which was then placed in a cooler (on ice) and kept cool until delivered to Maxxam in St. John's, NL. A new pair of nitrile gloves was used when collecting the sample.

6.2 Quality Assurance/Quality Control

Appropriate Quality Assurance/Quality Control (QA/QC) protocols were implemented.

Maxxam has an in-house QA program that consists of analyzing matrix spike, spiked blank and method blank samples. The results of these analyses are compared to established control limits to assess the quality of the analytical results. Maxxam is accredited with the Canadian Association for Laboratory Accreditation (CALA) and the Standards Council of Canada (SCC).

6.3 Soil Results

PCBs were not detected (<0.05 mg/kg) in the analyzed soil sample SA1A (0.3-0.8 m). The analytical results are not tabulated. However, the certificate of analysis is presented in Appendix C, for reference.

The laboratory QA report confirms the analytical results are within established tolerances and the data is considered to be representative. The QA report is presented with the certificate of analysis in Appendix C.

7.0 PCB DISTRIBUTION

The following presents a summary of the area(s) or locations where PCB exceedances exist or have been identified for the various sampled media.

7.1 Soil

There are two locations within the PCB Area where PCB concentrations in soil continue to exceed the CCME CSQG concentration of 33 mg/kg. The historical PCB results in soil are presented in Table 1, Appendix D. The two areas are referred to and described as follows:

Location A: PCB concentrations in this area range from 47.9 – 600 mg/kg and cover an area of approximately 23 m^2. This horizontal extent of PCB exceedances (i.e., >33 mg/kg) is considered to be well delineated. The vertical extent of impact is delineated at most of the sample locations within the impacted area. It is noted that several of the samples were collected at 3.0 m below surface (SA-1, SA-2, SA-3, SA-4, SA-5, SA-18, Dup-3 (a duplicate of SA-18)). Although impacted soil at these locations was excavated down to 3.0 m, it is assumed the PCB concentrations are also representative of the underlying soil. At these particular locations, the vertical extent of impact has not been delineated. The PCB impacted area is presented in Figure 4, Appendix A. Because of the number of soil samples that have been collected to date, Figure 4 only presents the analytical results for the sample locations where PCB concentrations exceeded the CCME CSQG concentration.

NH-TP-07: A single PCB concentration of 260 mg/kg in the soil sample NH-TP-07 exceeds the CCME CSQG concentration of 33 mg/kg. The impacted area at this location is estimated, recognizing there is only one sample with a PCB exceedance. The horizontal extent of PCB impacts is estimated to be approximately 6 m^2. The PCB impacted area is presented in Figure 4, Appendix A.

7.2 Groundwater

To date, PCB concentrations have not been detected in any of the analyzed groundwater samples with the exception of a PCB concentration of 0.07 ug/L detected in MW-01 in October 2009. However, PCB concentrations have not been detected in this monitoring well during three subsequent sampling events in 2010 and 2011. The historical groundwater results are presented in Table 2, Appendix D. The monitor well locations and the historical analytical groundwater results are presented in Figure 5, Appendix A.

7.3 Surface Water

To date, PCB concentrations have not been detected in the analyzed surface water samples with the exception of a PCB concentration of 0.05 ug/L detected in the Ditch sample in March 2009. It is noted that this concentration could not be confirmed on a certificate of analysis. The historical surface water

results are presented in Table 3, Appendix D. The surface water sample locations and historical analytical results are presented in Figure 6, Appendix A.

7.4 Sediment

To date, PCB concentrations have not been detected in the analyzed sediment samples, with one exception. A PCB concentration of 2.8 mg/kg was detected in the DITCH-SED sample collected in November 2010, which exceeds the CCME CSedQG (ISQG and PEL) concentration of 0.0341 mg/kg and 0.277 mg/kg, respectively. It is noted that that this sample location is located immediately adjacent to the PCB Area, and is more likely representative of the soil quality in the specific location. It is also noted that the sample was collected from a manmade interceptor/drainage ditch and does not necessarily fit the true definition of a sediment sample in the context of CCME CSedQG, as it is not considered to be an aquatic habitat.

The historical sediment results are presented in Table 4, Appendix D. The sediment sample locations and historical analytical results are presented in Figure 7, Appendix A.

7.5 Fish (Tissue)

PCB concentrations were not detected in the ten control and ten down-gradient fish samples. It is noted that the skin and the fat were removed from these samples before being analysed and current standard practice is to include the skin and fat as part of the analysis, as this is where PCBs will accumulate. The historical fish (tissue) results are presented in Table 5, Appendix D. The fish sample locations and historical analytical results are presented in Figure 8, Appendix A.

8.0 HUMAN HEALTH RISK ASSESSMENT

The human health risk assessment (HHRA) component of the overall HHERA is described in this section and its associated subsections. The ecological risk assessment (ERA) component of the HHERA is described in Section 9.0, and its associated subsections.

The HHRA of the PCB Area was conducted in accordance with the following Canadian regulatory guidance documents:

- *Federal Contaminated Site Risk Assessment in Canada, Part I: Guidance on Human Health Preliminary Quantitative Risk Assessment (PQRA), Version 2.0, September 2010. Prepared by: Health Canada Contaminated Sites Division, Safe Environments Directorate (Health Canada, 2010a);*

- *CCME Protocol for the Derivation of Environmental and Human Health Soil Quality Guidelines (CCME, 2006); and,*
- *Federal Contaminated Site Risk Assessment in Canada, Part V, Guidance on Human Health Detailed Quantitative Risk Assessment for Chemicals (DQRACHEM), September 2010. Prepared by: Health Canada Contaminated Sites Division, Safe Environments Directorate (Health Canada, 2010c).*

Where appropriate or necessary, the HHRA also considered human health risk assessment guidance and procedures developed and endorsed by regulatory agencies such as the United States Environmental Protection Agency (EPA), World Health Organization (WHO), Ontario Ministry of the Environment (OMOE), Alberta Environment and others.

The HHRA was also conducted in accordance with the GNLDEC 2005 Guidance Document for the Management of Impacted Sites (Version 1.01) and Policy Directive on the Management of Impacted Sites (i.e., PPD05-01). Both of these GNLDEC regulatory documents recognize risk assessment as an accepted practice in NL for managing contaminated sites. Within the Guidance document and Policy Directive, risk assessment frameworks and approaches developed by CCME and Health Canada are specifically identified as being acceptable to the GNLDEC. For sites that are complex or atypical, a risk-based approach (relative to a criteria-based approach) is often most appropriate or preferred, as such approaches can account for actual site-specific conditions (such as human and ecological receptor presence at a site, or use of a site by receptors; habitat quality; operable or complete exposure pathways; spatial extent of contamination, etc.).

Typically, a decommissioned waste disposal site is not subject to the Guidance Document and would not typically require a risk assessment. However, in the case of the PCB Area, there has been considerable public and regulatory interest over the years. Also, while remedial efforts have removed what is believed to be the majority of PCB-impacted soils, some residual PCBs remain in soil at the Site. Thus, in this situation, a risk assessment is an appropriate approach for assessing the residual PCBs in soil to determine if there is a human health and/or ecological risk.

8.1 HHRA Framework

The fundamental purpose of any HHRA is to estimate or determine whether people working, living at, or visiting a given location are being exposed, or are likely to be exposed to concentrations of chemicals that have the potential to result in adverse human health effects (i.e., toxicity). The potential for toxicity to occur as a result of chemical exposure is based on the dose response concept, which is fundamental to the responses of biological systems to all chemicals, whether they are therapeutic drugs, naturally occurring substances or anthropogenic chemicals in the environment. HHRA is a systematic tool or methodology that enables the prediction of people's exposure to chemicals in the environment and the potential health risks that may result from those exposures.

An HHRA typically consists of five main steps or phases, as outlined below and presented in Figure 8-1:

- **Problem Formulation**: identifies chemicals, receptors and exposure pathways/routes/scenarios of potential concern;
- **Exposure Assessment**: estimates exposure of hypothetical human receptors to each of the identified chemicals of potential concern (COPCs);
- **Toxicity (Hazard) Assessment**: determines an exposure limit or toxicological reference value (i.e., concentration of a chemical not expected to be associated with adverse health effects) for each of the COPCs;
- **Risk Characterization**: calculates risks associated with the estimated exposures and toxicity; and
- **Risk Management Recommendations:** determines the need for corrective action, and develops site-specific risk management objectives and/or criteria (such as site-specific target levels (SSTLs)), if required.

In Atlantic Canada HHRAs, it is relatively common for the term "quantitative human health risk assessment" to apply to the exposure assessment, toxicity assessment and risk characterization steps of HHRA.

The specific methods used to conduct each step of the HHRA, as well as the results or outcomes of these steps, are described and discussed in the following sections.

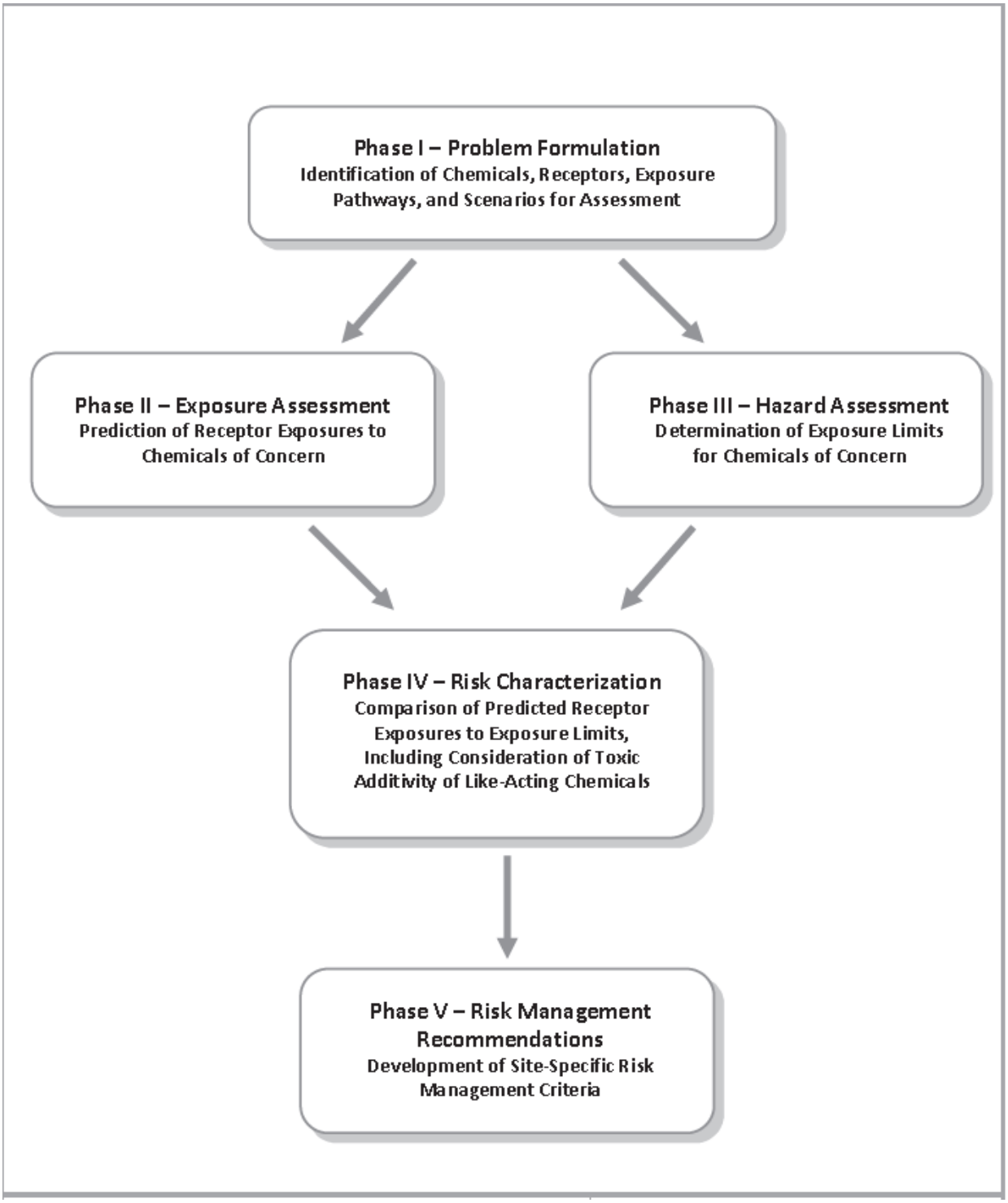


Figure 8-1 Five Phases of Human Health Risk Assessment

8.2 Problem Formulation

The problem formulation step is an important information gathering and interpretation stage, which serves to plan and focus the approach of the HHRA. The data gathered and evaluated in this stage provides information regarding the history and layout of the Site, possible exposure scenarios and pathways, identifies receptors of potential concern, identifies COPCs and any other specific areas or issues of concern to be addressed. Key tasks requiring evaluation within the Problem Formulation Step include the following:

- Characterize the Site;
- Selects Exposure Pathways and Routes;
- Identify COPCs;
- Select Exposure Scenarios;
- Select and Characterize Receptors; and
- Develops a Conceptual Site Model (CSM) that illustrates the key COPCs, receptors, exposure pathways and routes that will be evaluated in the HHRA.

The outcome of these tasks forms the basis of the approach taken in the HHRA. The methodology and outcomes for each of these tasks, as they relate to the PCB Area, are described in the following subsections, with the exception of Site Characterization, which was presented in Sections 3, 6 and 7.

8.2.1 Selection of Exposure Pathways and Routes

People can come into contact with chemicals in a variety of ways, depending on their daily activities and the land use (which is "commercial" for the Site). The means by which a person comes into contact with a chemical in an environmental medium are referred to as exposure pathways. The means by which a chemical enters the body from the environmental medium are referred to as exposure routes. There are three major exposure routes through which chemicals can enter the body: inhalation, ingestion and dermal absorption through the skin. For each of these major exposure routes, there are a number of potential exposure pathways. For example:

- Inhalation of gases, vapours, and dusts/particulate material through the lungs.
- Ingestion of soils, dusts, drinking water, garden produce, local food items (e.g., fish, shellfish, game meats, wild berries/plants), grocery store-bought food items; and accidental/incidental ingestion of surface water, ground water and sediments.
- Dermal absorption (uptake through the skin) from direct skin contact with gases/vapours, soils/dusts, water and other materials.

The potential for adverse health effects resulting from exposures to elevated chemical concentrations, in any medium, is directly related to the exposure pathways. If there is no pathway of exposure to a chemical, regardless of its toxic potency or concentration within a given medium, there is no potential for adverse health effects from that chemical to develop.

Given the Site's current and future commercial land use classification, the most likely and relevant human exposure pathways are direct contact with outdoor surface soil (via ingestion, dermal contact and soil/dust inhalation). As such, the following exposure pathways and routes were quantitatively evaluated in the HHRA:

- Inhalation of outdoor soils/dusts.
- Ingestion of outdoor soils/dusts.
- Dermal contact with outdoor soils/dusts.

Consideration was also given to evaluating the soil to outdoor air pathway for PCBs. While most commercial PCB formulations commonly used in electrical transformers (such as Aroclor products) are not highly volatile, trace amounts of PCBs have been documented to volatilize from soil at landfill sites (Environment Canada, 2001; ATSDR, 2000; WHO, 2003). The potential for volatilization of PCBs from soil to air is dependent on a number of conditions including the depth at which PCBs occur, the type of PCB mixture and its congener profile (for example, lower chlorinated congeners tend to have a higher vapour pressure than higher chlorinated congeners; the higher chlorinated congeners also tend to sorb more tightly to organic matter in soils and sediments), the age of the PCB mixture (as most organics have a decreasing potential for volatilization with increasing time in a soil matrix due to sorption, weathering, biodegradation and sequestration effects), various soil properties such as texture, organic carbon content, pH, temperature, moisture content, aeration/oxygenation, presence of microbial communities that can biodegrade or sequester PCBs, and the degree of ground vegetation cover. Overall, PCB soil to outdoor air volatilization rates are highly variable and are difficult to predict with any accuracy. When PCBs do migrate from soil to outdoor air, the lower molecular weight and/or less chlorinated congeners tend to occur primarily in the vapour phase, while the higher molecular weight and/or more chlorinated congeners tend to occur sorbed to airborne particulate matter (ATSDR, 2000).

While soil or groundwater to outdoor air exposure pathways do not typically require evaluation in HHRAs conducted in Atlantic Canada (as it is widely accepted that dilution and dispersion by winds and general atmospheric mixing prevents vapour concentrations from reaching levels that could cause harm), it is acknowledged that some Canadian jurisdictions (i.e., OMOE, 2011) do require evaluation of this pathway and have developed soil to outdoor air screening values for potentially volatile substances in soil, including PCBs. To date, no known jurisdictions have developed screening values based on the groundwater to outdoor air pathway (noting, PCBs have not been measurable in groundwater samples collected near the PCB Area or in other areas of the WDS to date).

As it is possible that trace amounts of PCBs could volatilize from soils within the PCB Area, and occur in outdoor air (as either vapours or particulate bound), the outdoor air inhalation pathway was considered in the HHRA. The outdoor air inhalation pathway is assessed qualitatively by comparing PCB Area soil concentration data to the applicable OMOE (2011) soil to outdoor air (S-OA) component values (discussed in Section 8.4). It should be noted that the inhalation of soils/dusts pathway (which is assessed quantitatively) already captures PCBs sorbed to soil particles that may become suspended or resuspended in outdoor air above the soils of the PCB Area, through such processes as wind erosion of soils or physical/mechanical disturbance of soils (e.g., digging, grading, vehicle traffic, etc.).

A number of other potential exposure pathways for the HHRA were considered, but were ultimately excluded from evaluation, as discussed in Table 8-1, below.

HHRAs commonly exclude pathways that are not relevant to the site or that lack sufficient data to enable their evaluation with a reasonable degree of confidence and/or accuracy. However, HHRAs are able to account for excluded pathways to some extent by adjusting (lowering) target hazard quotients (HQ) in the risk characterization step. This is described further in Section 8.3.4.

Table 8-1 Summary of Selected and Excluded Exposure Pathways in the HHRA

Exposure Pathway	Selected or Excluded?	Rationale
Inhalation of outdoor soils/dusts.	Selected	A likely exposure pathway.
Ingestion of outdoor soils/dusts.	Selected	A likely exposure pathway.
Dermal contact with outdoor soils/dusts.	Selected	A likely exposure pathway.
Inhalation of vapours/particulates in outdoor air	Selected	A possible (albeit unlikely to be significant) exposure pathway.
Groundwater ingestion (as drinking water) and dermal contact with groundwater	Excluded	Groundwater from the vicinity of the PCB Area and the rest of the WDS is not used, nor was it ever used, for potable purposes. There are no water supply wells on any portion of the PCB Area or WDS. No water supply wells are expected on any portion of the WDS following its closure. PCBs have not been detected in groundwater samples collected from the vicinity of the PCB Area or other areas of the WDS. Furthermore, PCBs tend to strongly sorb to soils and generally do not leach to groundwater to any significant extent (due to low water solubility and generally high affinity of PCB congeners for organic carbon in soils) (ATSDR, 2000). Landfill soils (especially those that received mixed residential and commercial waste, as is the case for the WDS) tend to have elevated organic carbon concentrations. An exception to the generally low leaching potential for PCBs may occur if the PCBs are present in soil with

Exposure Pathway	Selected or Excluded?	Rationale
		organic solvents as co-contaminants. Where this occurs, the PCBs may dissolve in the solvents and more easily undergo vertical migration to groundwater, as most solvents have higher water solubility than PCBs. This does not appear to be the case at the PCB Area or other portions of the WDS as historical soil and groundwater monitoring results have been largely non-detectable for VOCs (which includes solvents). Where such substances have been measurable, the soil and groundwater concentrations have been low.
Indoor dust ingestion, inhalation and dermal contact	Excluded	Lack of current site buildings/structures and no expected buildings/structures for future land use. Thus, these exposure pathways are incomplete.
Indoor vapour inhalation	Excluded	Lack of current site buildings/structures and no expected buildings/structures for future land use. Thus, this exposure pathway is incomplete.
Ingestion of, and dermal contact with water and sediments from local surface water bodies	Excluded	No expected contact for site visitors with surface water bodies, as these features are located off-site. Monitoring of PCBs in streams and ponds near the PCB Area has indicated no measurable PCB concentrations to date in either surface water or sediments.
Ingestion of locally caught or harvested foods	Excluded	Neither the PCB Area nor the rest of the WDS will be used for any agricultural or garden activities. No wild berries are present in the vicinity of the PCB Area or the rest of the WDS. The PCB Area and other portions of the WDS will not likely be used for hunting or trapping purposes. Available fish fillet concentration data for fish collected from water bodies down-gradient of the PCB Area (and in direction of groundwater flow) has indicated non-detectable PCB concentrations. Furthermore, PCBs are not readily accumulated by terrestrial plants (ATSDR, 2000; Environment Canada, 2001). Reported bioaccumulation factors (ratios between soil and plant PCB concentrations) for a number of plant species consumed by humans is much lower than 1.0.
Market basket (grocery store) food items ingestion	Excluded	Potential exposures to PCBs in purchased food items would be no different for individuals accessing the PCB Area of the WDS than it would be for those individuals that do not access the Site.

In summary, the following exposure pathways were evaluated in this HHRA:

- Inhalation of outdoor soils/dusts.
- Ingestion of outdoor soils/dusts.
- Dermal contact with outdoor soils/dusts.
- Inhalation of outdoor air.

8.2.2 Identification of Chemicals of Potential Concern (COPCs)

As previously stated in Section 2.0, this HHERA (of which the HHRA is a component), is limited to the evaluation of PCBs only. Thus, PCBs are the sole COPCs for the HHRA.

All available media and biota chemistry data relating to the PCB Area of the WDS is expressed as total PCBs. No data has been collected to date on individual PCB congeners (of which there are 209 distinct substances), including the coplanar congeners, which exhibit dioxin-like activity in biological systems and are evaluated in the same manner as polychlorinated dibenzo-*p*-dioxins and dibenzofurans (i.e., using toxic equivalency factors and total toxic equivalents or TEQ). Soil and other media chemistry data for total PCBs (when detectable) typically includes resemblance data for the PCB technical or commercial formulations, similar to petroleum hydrocarbon product resemblance data that are routinely reported by laboratories in Atlantic Canada. The PCB formulations used in transformers within North America were complex mixtures of individual PCB congeners, known by the trade name "Aroclor". Trade names used for the same or very similar formulations in other countries included Clophen, Kanechlor, Fenclor, Phenchlor. Relatively few Aroclor mixtures were used as transformer fluids in Canada. For the PCB Area, the available soil chemistry data indicates PCBs originated from Aroclor 1254, Aroclor 1260 and Aroclor 1242 transformer fluid products. Aroclors are identified by a four-digit numbering code in which the first two digits indicate the type of mixture and the last two digits indicate the approximate chlorine content by weight percent (ATSDR, 2000).

8.2.3 Selection of Exposure Scenarios

Based on information previously presented in relation to site characterization, COPC selection, and exposure pathway identification, the primary exposure scenario considered in the HHRA is the Current Scenario (As Is - Outdoor Site Visitor (Trespasser)).

The Site (including the PCB Area) is currently classified as commercial, which is expected to continue for the foreseeable future. There will be no buildings or infrastructure within the PCB Area other than the engineered cap. While access to the PCB Area is somewhat restricted by fencing along the road, there are no major barriers that would prevent or greatly restrict human access to the PCB Area of the WDS.

This exposure scenario was evaluated quantitatively (with the exception of outdoor air exposure, which was assessed on the basis of benchmark comparisons) and assumes an outdoor site visitor or trespasser could incur potential exposures to PCBs via outdoor air inhalation and direct soil contact pathways that include soil ingestion, dermal contact, and inhalation of soil/dusts while on the Site.

There is no firm or consistent guidance within North American regulatory agencies on exposure frequency (EF) and exposure duration (ED) assumptions for infrequent or transient site access scenarios such as that evaluated herein. Rather, professional judgement is typically used to estimate reasonable worst case conditions for exposure frequency and duration, for people that may access the Site. Health

Canada (2009) previously endorsed an assumption of 2 days/week, 35 weeks/year for recreational land use. However, this land use category and its corresponding assumptions were dropped when the final Health Canada HHRA guidance was published (i.e., Health Canada, 2010a). As the PCB Area will remain as a commercial land use designation (even though there will be no future commercial activity), and is unlikely to be preferentially used for recreational purposes relative to any other local area that has trails, woods, ponds and streams (of which there are many), the previous Health Canada (2009) assumptions for EF and ED are considered unrealistic to apply to the Site. Thus, for the purposes of this HHRA, the assumed EF and ED values for the Current Scenario (As Is - Outdoor Site Visitor (Trespasser)) are based on professional judgement, and are: 3 hours/day, 1 day/week, 48 weeks/year. Appendix E provides a sample calculation illustrating how exposures and risks were calculated in the HHRA.

The Current Scenario (As Is - Outdoor Site Visitor (Trespasser)) is based on current residual surface soil concentrations of PCBs within the PCB Area.

Because the closure plan for the WDS will include placing an engineered cap over the PCB Area, a secondary exposure scenario was also evaluated to determine if the cap is likely to minimize potential human and ecological exposures and risks. This Future Scenario (Post-Capping) is evaluated qualitatively based on such considerations as depth of the cap layers, the types of cap layers, the outcomes of the primary HHRA scenario, and review of all the environmental media and biota chemistry data that has been collected to date.

The assessment of exposures that may occur as part of future Site development scenarios is not within the scope of this current HHRA as there are no known plans for re-development of the Site or any other portion of the WDS at this time.

The evaluation of potential exposures and risks to workers who may participate in Site decommissioning work within the PCB Area at some point in the future (e.g., construction or utility workers), is also beyond the scope of this HHRA. It is understood that the GNLDEC will require an H&S Plan be developed for the installation of the engineered cap over the PCB Area, or for any subsequent excavation/maintenance work within the PCB Area.

Spatial boundaries for both of the assessed exposure scenarios are the current areal extent of the PCB Area (i.e., approximately 0.0913 ha), and the off-site monitoring locations that have been used for evaluating whether or not PCBs are migrating from the PCB Area to the down-gradient water courses and water bodies. No other properties or off-site areas are considered in the HHRA. Temporal boundaries are current conditions for the Current Scenario (As Is - Outdoor Site Visitor (Trespasser)) and potential future conditions for the Future Scenario (Post-Capping).

8.2.4 Receptor Identification and Characterization

A human receptor is a hypothetical person (e.g., infant, toddler, child, adolescent or adult) who resides, visits or works in the area being assessed and is, or could potentially be, exposed to the chemicals identified as being of potential concern. General physical and behavioural characteristics specific to the receptor type (e.g., body weight, breathing rate, amount of soil and food consumed, etc.) are used to determine the amount of chemical exposure received by each receptor. Due to differences in these characteristics between children and adults and between males and females, the exposures received by a female child, a male child, a female adult or a male adult will be different. Consequently, the potential risks posed by the chemicals being evaluated will also differ depending on the receptor chosen for evaluation.

Since people have varying physical features, lifestyles and habits, it is not possible to evaluate all types of individuals. However, an HHRA must be sufficiently comprehensive to ensure that those receptors with the greatest potential for exposure to chemicals of concern, and/or those that have the greatest sensitivity or potential for developing adverse effects from such exposures are included in the evaluation. If no potential health risks are determined for relevant receptors which are considered to be either the most sensitive, or receive the greatest exposures from a given site, then it can be assumed that those receptors who are either less sensitive, or who receive lower exposures, would also not be at risk.

A female toddler receptor was selected for evaluation in the HHRA. In HHRAs, it is commonplace to select toddler receptors for the evaluation of chemicals not considered to be carcinogens (i.e., threshold response chemicals). Current Health Canada toxicity reference values for PCBs are based on non-carcinogenic effects (i.e., Health Canada, 2010b). While it is considered unlikely that toddlers would spend time at or in the vicinity of the PCB Area, toddlers are a conservative and protective choice for a human receptor as they typically receive greater chemical exposures via all pathways and routes, on a relative body weight basis when compared to other human receptor classes (i.e., male toddler, child, adolescent and adult). Similarly, as females tend to weigh less than males, a female toddler tends to incur higher chemical exposures than a male toddler, on a body weight basis. In addition, toddlers (both females and males) have certain physiological and behavioural characteristics that tend to increase their chemical exposure relative to other receptors (e.g., higher chemical intake rate to body weight ratios; tendency to play outdoors for prolonged durations; tendency to ingest soil/dust due to frequent hand-to-mouth behaviour, mouthing of objects; greater potential for dermal contact due to playing and digging in soil, etc.).

While Health Canada HHRA guidance in recent years has moved away from distinguishing between genders for the selection of human receptors, HHRA guidance in some other jurisdictions does continue to differentiate receptors based on gender. Although differences in body weights, intake rates, etc., are generally minor between males and females, assessing a female receptor is inherently more conservative,

due to the slightly higher intake rate to body weight ratios, relative to male receptors. Also, for a number of chemicals, there is toxicological evidence that females may be more sensitive than males.

Given the above considerations, selecting a female toddler receptor is a conservative approach for the HHRA that is protective of the other human receptors (i.e., male toddler, children, adolescent or adult).

As the human carcinogenicity of PCBs is currently not conclusive, and Health Canada has not developed cancer-based toxicity reference values for PCBs to date (discussed further in Section 8.3.2), PCBs were treated as non-carcinogens in this HHRA. As such, it was not necessary to evaluate a composite or lifetime receptor in the HHRA. When chemicals are considered to be carcinogens (such as arsenic, certain polycyclic aromatic hydrocarbons (PAHs) and benzene) are assessed in HHRAs, it is necessary that such substances be evaluated over an appropriate period of time, as the development of cancer is a long term process that may take many years to manifest. For this reason, a special type of hypothetical receptor called a "lifetime" or "composite" receptor is evaluated for potential carcinogenic risks. A female lifetime receptor is often chosen since females generally incur higher exposures than males based on their lower body weight, as discussed above. Assessment of lifetime receptors in HHRAs involves combining (using a weighted average approach) estimated daily exposures for each relevant life stage, to calculate a Lifetime Average Daily Dose (LADD), which is then used to estimate incremental lifetime cancer risk (ILCR).

Key parameters and assumptions describing the physiological and behavioural characteristics of the hypothetical human receptor evaluated in the HHRA (as well as some key environmental parameters) are presented in Table 8-2. Appendix E presents an example calculation illustrating how the receptor and environmental parameters are used within the HHRA calculations.

Table 8-2 Key Receptor and Environmental Parameters and Assumptions for the Female Toddler Receptor

Parameter	Value	Reference
Body weight (kg)	16.4 (arithmetic mean for Canadian population)	Richardson, 1997
Inhalation rate (m^3/day)	8.3	Health Canada, 2010a
Duration of life stages modelled (yrs)	4.5	Health Canada, 2010a
Time spent on-site (exposure frequency (EF) and exposure duration (ED))	3 hours/day, 1 day/week, 48 weeks/year However, as PCBs are a known developmental toxicant (ATSDR, 2000; WHO, 2003), the exposure amortization (for EF and ED) was reduced to 1 day/week (as per Health Canada, 2010a guidance for exposure time amortization for both event driven soil contact exposure and substances that are known	Assumed based on professional judgement

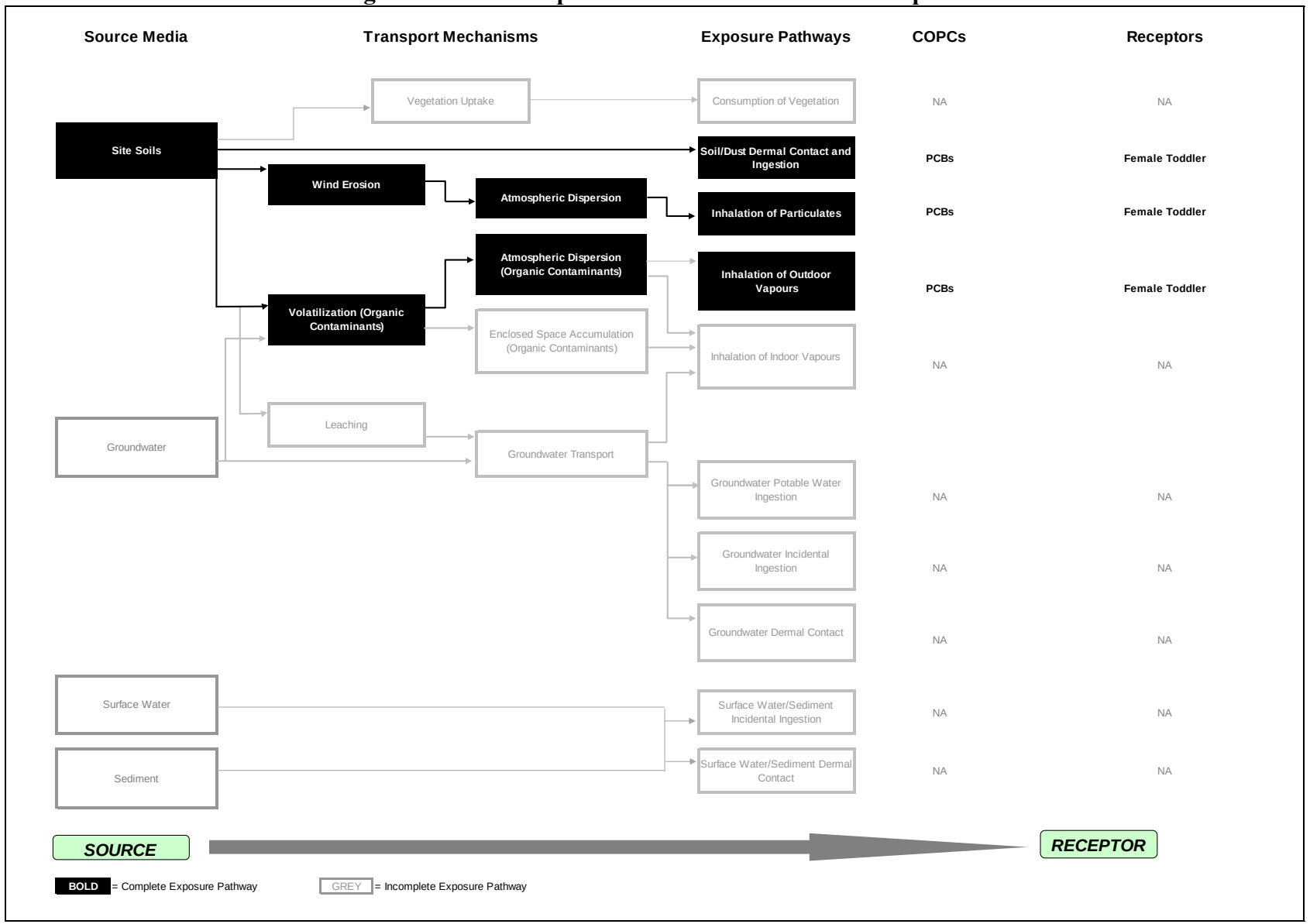
Parameter	Value	Reference
	to be developmental toxicants)	
Soil ingestion rate (g/d)	0.08	CCME, 2006; Health Canada, 2010a
Hand surface area (m^2)	0.043	Richardson, 1997; Health Canada, 2010a; CCME, 2006
Hand soil adherence factor ($g/m^2/d$)	1	CCME, 2006; Health Canada, 2010a
Skin surface other than hands (m^2) (upper and lower arms and legs)	0.277	Richardson, 1997
Skin soil adherence factor ($g/m^2/d$)	0.1	CCME, 2006; Health Canada, 2010a
Outdoor dust level from soil (g/m^3)	7.6×10^{-7} (recommended Health Canada, 2010a default value for urban dust level)	Health Canada, 2010a
Fraction of airborne dust generated from site soil	1.0	Assumed
Winter cover factor	0.67 (fraction of days without winter cover)	Based on EC Climate Normals Data For Heart's Content and Holyrood Generating Station meteorological stations (1971-2000). Both stations are located in similar terrain and coastal proximity as the Site, and are the closest available stations to the Site. Heart's Content station is approximately 33 km north of the Site, while the Holyrood station is located approximately 33 km southeast of the Site. (http://www.climate.weatheroffice.ec.gc.ca/climate_normals/index_e.html) Consideration was also given to Environment Canada's 'Climate Data Online' for NL stations. (http://www.climate.weatheroffice.gc.ca/climateData/canada_e.html This database has more recent records spanning 2001 to 2011, though not necessarily for the same data categories and locations as the Climate Normals Data. Overall, it was considered that months with average snowfall >20 cm, and/or daily average or daily max temperatures of <0 degrees C, and/or months with snow depth >10 cm, are winter months. It is assumed that under these weather conditions, snow or ice cover is likely, and people accessing the Site would have little to no exposed skin surface, such that soil contact exposure pathways are negligible.

It is considered that the most likely users of the Site (and PCB Area) will be residents of local communities who may occasionally hike, walk or ride ATVs in the vicinity of the WDS, or occasionally pass through the WDS to access nearby off-site areas for hunting, snaring, fishing and berry picking activities. The receptor parameters presented in the above table capture these potential Site access/use scenarios and are considered to substantially overestimate the potential PCB exposures that might occur under these access/use scenarios.

8.2.5 HHRA Conceptual Site Model

The HHRA conceptual site model (CSM) is presented in Figure 8-2. The CSM provides a simplified representation of potential exposure pathways that link the identified COPCs to the human receptors of interest.

Figure 8-2 Conceptual Site Model for Human Receptors



8.3 Qualitative Human Health Risk Assessment

8.3.1 Exposure Assessment

Exposure assessment involves estimating the amount of chemical received by individuals per unit time (i.e., the quantity of chemical and the rate at which that quantity is received). The exposure assessment evaluates data related to all chemicals, human receptors and exposure pathways selected during the problem formulation step of the HHRA. The rate of exposure to chemicals from the various environmental media being evaluated is usually expressed as a dose, or the amount of chemical taken in per body weight per unit time (e.g., µg chemical/kg body weight/day).

The degree of human exposure to chemicals in the environment depends on the interactions of a number of parameters, including:

- The concentrations of chemicals in various environmental media (e.g., air, water, soil, food) as determined by the quantities of chemicals entering the environment from various sources, their fate and persistence in these media, and the normal ambient or background concentrations that are present independent of a specific source.

- The various exposure pathways for the transfer of the chemicals from the different environmental media to humans (e.g., inhalation of indoor and outdoor air, soil particles and dusts; ingestion of food items, water, soils/dusts; skin penetration of various chemicals from swimming or dusts landing on skin).

- The physiological, behavioural and lifestyle characteristics of human receptors that determine the actual exposures through interactions with the various pathways (e.g., respiration rate, water intake, food intake, soils/dusts intake, time spent at various activities and in different locations).

- The various physical, chemical and biological factors that determine the ability of people to take the chemicals into their bodies from the exposure pathways (e.g., bioavailability of the chemicals from soil/dust particles, foods, water, and air).

For this HHRA, deterministic (point estimate) exposure analysis was conducted. Typically, HHRAs determine a reasonable upper bound point estimate exposure for all receptors and exposure scenarios evaluated. This helps identify areas where more detailed approaches (such as further data collection or the use of probabilistic modelling techniques) could/should be used to refine assumptions to enable a more site-specific and realistic exposure assessment. However, in situations where the upper bound point estimate approaches indicate no potential risks to receptors, or risk management decisions based on such approaches are considered reasonable, it is often unnecessary to apply more detailed/sophisticated site-specific exposure and risk assessment approaches.

Characterization of Site Soil Data for Exposure Assessment

As previously stated, soil is the only Site medium that contains detectable concentrations of residual PCBs, and as such, is the focus of the HHRA.

Various summary statistics were calculated for the PCB soil concentrations, including range, arithmetic mean, upper 95% confidence limits on the arithmetic mean (UCLM95) and upper percentiles. Of these, the most important summary statistic is the UCLM95. Accurate and statistically robust estimates of the UCLM95 are important in an HHRA as most regulatory agencies recognize the best approximation of the exposure point concentration (EPC), often referred to as the "true mean" or "the concentration most likely to be contacted over time", is the 95% upper confidence limit (UCL) of the arithmetic mean. To derive the UCLM95, the U.S. EPA computer program, ProUCL™ Version 4.1 was used. The U.S. EPA recommends the use of ProUCL when calculating EPCs for use in HHRAs of contaminated sites. ProUCL determines the most appropriate UCLM95 value for a dataset, given its distribution and characteristics. A number of statistically valid methods to calculate a UCLM95 can be run simultaneously, with the program recommending options for the most appropriate or statistically robust value(s) to select. However, according to its user guidance, ProUCL can only determine robust and reliable UCLM95 values if the sample size is at least 8. As the sample size within the Site soil chemistry dataset is much larger than this (i.e., N = 64 for surface soil PCB data; and, N = 127 for combined surface and subsurface soil PCB data), it was possible to calculate an adequately robust UCLM95 soil concentration for PCBs. In calculating the UCLM95 soil concentration of PCBs (i.e., the EPCs), the following tasks/conditions were conducted/applied:

- The HHRA used the analytical data from the previous ESA, remediation and monitoring programs conducted since 2003. Dillon carefully reviewed the soil sample locations, to determine which ones were representative of current soil conditions. Several soil sample locations with corresponding analytical results were eliminated, as the soil at these locations was subsequently excavated and disposed off-site during one of the remediation programs. As such, the analytical results that were evaluated as part of the HHRA are considered to be representative of the current soil quality.
- The soil depth of the residual PCB soil concentrations was a key consideration. Given the anticipated future land use of the Site, and the most likely human access/use scenarios (i.e., occasional site visits or trespassing for recreational purposes), only surficial soil depth is relevant towards evaluating potential human exposure to PCBs in soil. For consistency with the standard CCME surface/subsurface soil depth cut-off (i.e., 1.5 m), it was assumed that potential surface soil PCB concentrations were represented by PCB soil sample result that corresponded to a depth of less than or equal to 1.5 m. PCB soil sample results from depths of greater than 1.5 m were considered to be unavailable for potential human exposure (with the exception of the pathway of PCB volatilization to outdoor air and subsequent outdoor air inhalation; for this pathway, all surface and subsurface residual PCB soil concentration data were used).

- For soil samples with corresponding laboratory or field duplicates, the higher PCB concentration of the original and duplicate sample was retained for the purpose of calculating the EPC.
- For soil sample results where PCBs were not detectable (i.e., <Reportable Detection Limit (RDL)), the <RDL values were assumed to equal the RDL for the purposes of EPC calculation.
- Prior to calculating the EPC for soil PCB concentrations, Dillon reviewed the laboratory certificates of analysis for the soil sample submissions, as well as the accompanying laboratory quality assurance reports. This review focused on laboratory performance with respect to the RDLs that were achieved, % surrogate recoveries, lab and field duplicate results and relative percent difference (RPD) or absolute difference (when lab duplicates are compared to original sample results), matrix spikes, method blanks, and spiked blanks. While there were a few instances where soil samples had elevated RDLs, overall, no major analytical issues were identified that would affect the use of these data in an HHRA. Thus, the soil PCB chemistry data were considered to be of adequate quality for use in an HHRA. Further information on soil data QA/QC is provided within many of the previous ESA, remediation and monitoring reports.
- As the PCB concentrations in soil samples represent potential concentrations that human receptors could come into contact with, no attempt was made to conduct statistical outlier tests to remove extreme values (high or low) from the soil chemistry dataset. Thus, the EPC calculations included the presence of potential extreme values.
- As the calculated options for a UCLM95 generated by ProUCL 4.1 can vary considerably (as a function of the underlying assumptions in the statistical models and the soil data distribution type), some degree of professional judgement is typically necessary in selecting the most appropriate UCLM95 value for use as the EPC in a HHRA. Key considerations often include the data distribution type, the significance level associated with the UCLM95 calculation methods (i.e., ProUCL-recommended values are not always at the 95% significance level, nor do they always correspond to the underlying distribution type), any warnings generated by the ProUCL 4.1 software and the magnitude of the calculated UCLM95 options.

The calculated EPCs for soil PCB data that were evaluated in the HHRA are presented in Table 8-3 below, along with a brief description and rationale for the selected EPC values. The ProUCL 4.1 output that was used to determine the EPCs is presented as an attachment to Appendix F.

Table 8-3 Exposure Point Concentrations (EPCs) for PCBs in Soil

PCB Data	Selected EPC Value (UCLM95)	Description/Rationale
Surface soil residual PCB concentrations only (i.e., ≤1.5 m); N = 64	27.1	95% Chebyshev (Mean, Sd) UCL; soil data do not follow a discernible distribution, thus, a non-parametric statistical model is most appropriate; the selected value is the most reliable and appropriate value out of the options calculated by ProUCL 4.1 software.
Combined surface and subsurface soil residual PCB concentrations; N = 127	51.0	95% Chebyshev (Mean, Sd) UCL; soil data do not follow a discernible distribution, thus, a non-parametric statistical model is most appropriate; the selected value is the most reliable and appropriate value out of the options calculated by ProUCL 4.1 software.

Notes:
UCLM95 and UCL both indicate an upper 95% confidence limit on the arithmetic mean.

Exposure Equations

The following equations (based on those provided within Health Canada, 2010a), were used to estimate exposures to PCBs in soil in the HHRA, for the pathways that were evaluated in a quantitative manner (i.e., soil/dust inhalation, ingestion and dermal contact). A winter cover factor (i.e., 0.67; see Section 8.2.4) was applied to the total exposure for each relevant pathway(s), prior to calculating the risk estimates. Appendix E presents an example calculation illustrating how exposures and risks were calculated in the HHRA.

Outdoor Soil/Dust Ingestion Exposure (μg/kg body weight/day) = [SIR x Cs x AF x ET]/ BW

Where:

SIR = soil ingestion rate (g/day)
C_s = soil chemical concentration (μg/g)
AF = fraction of chemical absorbed *via* soil ingestion (chemical-specific; unitless; for PCBs, this value is 1.0 as described in Section 8.3.3)
ET = exposure time (unitless); ET is calculated from the assumed exposure frequency and duration (i.e., 1 day/week = 0.14; see Table 8-2)
BW = body weight (kg)

Outdoor Inhalation of Soil/Dust Exposure (μg/kg body weight/day) = [AI x C_s x AF x DL x ET x FDS]/BW

Where:

AI = amount of air inhaled (m^3/day)
C_s = soil chemical concentration (μg/g)
AF = fraction of chemical absorbed *via* inhalation (chemical-specific; unitless; for PCBs, this value is 1.0 as described in Section 8.3.3)

DL = outdoor dust level (g/m^3)
ET = exposure time (unitless); ET is calculated from the assumed exposure frequency and duration (i.e., 1 day/week = 0.14; see Table 8-2)
FDS = fraction of dust generated from site soil = 1.0 (Assumed; 100% of the airborne dust is assumed to come from the site soils, and is not influenced by other typical ambient dust sources)
BW = body weight (kg)

Outdoor Dermal (Hand) Soil/Dust Exposure (µg/kg body weight/day) = [HSA x HSAF x C_s x AF x ET]/BW

Where:

HSA = hand surface area (m^2)
HSAF = hand soil adherence factor (g/m^2/day)
C_s = soil chemical concentration (µg/g)
AF = fraction of chemical absorbed *via* dermal contact (chemical-specific; unitless; for PCBs, this value is 0.14 as described in Section 8.3.3)
ET = exposure time (unitless); ET is calculated from the assumed exposure frequency and duration (i.e., 1 day/week = 0.14; see Table 8-2)
BW = body weight (kg)

Outdoor Dermal (Arms and Legs) Soil/Dust Exposure (µg/kg body weight/day) = [SSA x BSAF x C_s x AF x ET]/BW

Where:

SSA = skin surface area (m^2)
BSAF = body soil adherence factor (g/m^2/day)
C_s = soil chemical concentration (µg/g)
AF = fraction of chemical absorbed *via* dermal contact (chemical-specific; unitless; for PCBs, this value is 0.14 as described in Section 8.3.3)
ET = exposure time (unitless); ET is calculated from the assumed exposure frequency and duration (i.e., 1 day/week = 0.14; see Table 8-2)
BW = body weight (kg)

8.3.2 Toxicity (Hazard) Assessment

Toxicity is the potential for a chemical to produce any type of damage, permanent or temporary, to the structure or function of part of an organism. The toxicity of a chemical depends on the amount of chemical taken into the organism (referred to as the "dose") and the duration of exposure (i.e., the length of time the person or organism is exposed to the chemical). For every chemical, there is a specific dose and duration of exposure necessary to produce a toxic effect in an organism (this is referred to as the "dose-response relationship" of a chemical). In the toxicity assessment step of an HHRA, information relating to the dose-response relationship of each COPC is evaluated (usually from laboratory animal

studies and studies of human exposure in the workplace) in order to determine the maximum dose to which humans can be continuously exposed, with no or a very low probability of experiencing adverse health effects. These toxicity estimates are called toxicity or toxicological reference values (TRVs) (also referred to as exposure limits, the terms are analogous) and indicate an exposure that will not likely result in adverse human health effects.

TRVs are typically derived by regulatory agencies based on detailed reviews of toxicological, epidemiological and other scientific information, professional judgement and technical review by a number of experienced scientists with expertise in the toxicological sciences. They are often derived based on the most sensitive endpoints in individuals (e.g., organ damage, neurological effects, cancer, reproductive effects, etc.) and large safety or uncertainty factors (i.e., 100-fold or greater) are commonly used in their estimation. These factors are often applied to exposure levels from studies where no adverse effects were observed (i.e., the no observed adverse effects level (NOAEL)). Thus, exceedance of a TRV does not necessarily mean that adverse effects will occur; rather, it means the safety factor beyond the no-effect exposure is somewhat reduced. Usually, exposure rates that are less than TRVs are not likely to be associated with adverse health effects, and are therefore, less likely to be of concern. As the frequency or magnitude of exposures exceeding a TRV increase, the probability of adverse health effects occurring in a human individual or population also increases. However, it should not be categorically concluded that all exposures below the TRV will be acceptable (unlikely to result in adverse health effects) and that all exposures above the TRV are unacceptable (likely to result in adverse health effects).

There are two main types of dose-response relationships that have been established for chemical agents:

- **Threshold Response Chemicals**: For these substances, there is a dose-response threshold below which no adverse effects would be expected to occur. This relationship is generally true for all chemicals that do not cause cancer by altering genetic material. Thresholds are generally assumed to exist for non-carcinogens because for non-carcinogenic effects, it is believed that homeostatic, compensating and adaptive mechanisms must first be overcome before toxicity is manifested. TRVs derived for threshold-response chemicals are often referred to as reference doses (RfD), acceptable daily intakes (ADI), tolerable daily intakes (TDI) or permissible daily intakes (PDI) and are generally derived by regulatory agencies. These values indicate concentrations of chemicals that individuals can be exposed to on a continuous basis without the occurrence of adverse health effects. TRVs derived for threshold-response chemicals are typically expressed as either a dose (e.g., µg/kg body weight/day) or a media concentration (e.g., $\mu g/m^3$).
- **Non-threshold Response Chemicals**: For these types of chemicals, it is assumed there is no dose-response threshold. This means that any exposure greater than zero is assumed to cause some type of response or damage. This relationship is typically used for chemicals which can cause cancer by damaging genetic material. Since, in theory, any exposure has the potential to cause damage, it is necessary to define an "acceptable" degree of risk associated with these types of exposures. This

"acceptable" degree of risk is usually defined as a risk of one-in-one hundred thousand to one-in-one-million. These numbers can be better explained as the dose rate that may cause an increased cancer risk in one person out of one hundred thousand people, or one person out of one million people. The acceptable level of carcinogenic risk is a policy rather than a scientific decision, which is set by regulatory agencies as opposed to risk assessors. In the Atlantic Provinces, and for sites owned by the Canadian federal government, an acceptable target cancer risk level of 1 in 100,000 ($1e^{-5}$) has been established. TRVs derived for non-threshold chemicals believed to be potential carcinogens are typically expressed as cancer slope factors or cancer potency factors [e.g., $(\mu g/kg \text{ body weight/day})^{-1}$)] or unit risk values for environmental media [e.g., $(\mu g/m^3)^{-1}$]. However, TRVs for carcinogens may also be expressed as risk-specific media concentrations or doses associated with a particular level of acceptable cancer risk.

The assumption of no dose-response threshold for carcinogens ignores a large number of factors, such as the ability of the body to repair damage to genetic material, that are known to be important in the responses of people to naturally occurring genotoxic carcinogens. Exposure to small concentrations of chemicals which have the potential to cause cancer happen on a daily basis to everyone in the world, because these chemicals (along with many other chemicals which do not cause cancer) are continuously present in all environmental media and food items, either from natural sources or due to human activities. The human body has many ways of handling these substances once they are absorbed, and in many cases, can repair the damage that may be caused by exposures to low levels of potentially carcinogenic chemicals.

In this HHRA, oral and inhalation TRVs for the chemicals identified as being of potential concern (i.e., PCBs in this case) were identified from such reputable regulatory sources as: Health Canada, U.S. EPA, the California EPA Office of Environmental Health Hazard Assessment (CalEPA OEHHA), the U.S. Agency for Toxic Substances and Disease Registry (ATSDR), the National Institute for Public Health and the Environment (RIVM), Netherlands, and the WHO. In selecting the TRVs, careful consideration was given to such aspects as scientific basis (e.g., health endpoint and toxicological basis, degree of conservatism in their derivation, quality of principal study, presence or absence of supporting scientific data from other studies, application of uncertainty or modifying factors, derivation method used), date of last major review or update and the speciation (or chemical form, formulation, etc.) of the chemical of interest in the principal study, if known. The TRVs selected for this HHRA (See Table 8-4) are routinely used in HHRAs conducted in Canada and elsewhere, and are protective of chronic exposure conditions. For the most part, TRVs are developed to be protective of all members of a human population including sensitive life stages (such as the elderly, pregnant women) and individuals of compromised health, where data allow.

There are presently no regulatory dermal TRVs available for PCBs that are expressed as a dose or cancer slope factor. Thus, dermal exposures were accounted for in the HHRA by using a relative bioavailability

approach (See Section 8.3.3, below). A number of regulatory agencies, including Health Canada (2010a) and the U.S. EPA (numerous guidance documents prepared within the Superfund Program), provide guidance for considering relative bioavailability between the dermal and oral exposure routes using dermal relative absorption factors (RAFs) and oral TRVs.

Table 8-4 Summary of TRVs Considered and Used in the Human Health Risk Assessment

Chemical	Exposure Route	Chronic Toxicity Reference Value (TRV)		Health Endpoint(s)	Principal Study(ies)	Regulatory Agency Source
		Type	Value			
PCBs	Oral	TDI	0.13 µg/kg body weight/day	NOAEL for neurodevelopmental effects in monkeys exposed to Aroclor 1248; 100-fold UF applied	Bowman et al., 1981	Health Canada, 2010b
	Oral	RfD; TDI; MRL	0.02 µg/kg body weight/day	LOAEL for immunological effects, liver weight increases, and mild dermal and ocular effects in monkeys exposed to Aroclor 1254; 300-fold UF applied	Arnold et al., 1993a,b; Tryphonas et al., 1989; 1991a,b	U.S. EPA, 1996; ATSDR, 2000; WHO, 2003
	Inhalation	TCA	0.5 µg/m^3	LOAELs in various experimental animals exposed to Aroclor 1254 (details not provided by source agency); the TCA is one half of the values obtained for Aroclor 1254 to account for potential exposure to key PCB congeners of concern in other PCB mixtures	Details not provided by source agency	RIVM (Baars et al., 2001)

Notes:
TDI = tolerable daily intake; NOAEL = no observable adverse effect level; UF = uncertainty factor; RfD = reference dose; MRL = minimal risk level; TCA = tolerable concentration in air; LOAEL = lowest observable adverse effect level.

As presented in Table 8-4, the inhalation TRVs for PCBs have an uncertain basis due to limited documentation provided by the source agency (RIVM in the Netherlands in this case). ATSDR (2000) notes that there is generally a lack of adequate human and animal inhalation toxicity data for PCB mixtures. A brief search of the Toxline (with Medline) literature database indicated while some studies on PCB inhalation exposures and/or toxicity have been published since the ATSDR profile was last updated and since the RIVM TCA was developed, there does not appear to be sufficient studies to support development of an alternate inhalation TRV at this time for any PCB mixture. The Tolerable Concentration in Air (TCA) of 0.5 µg/m^3 does appear however, to be a protective value based on review of the NOAELs and LOAELs (expressed as air concentrations) that are reported within the available subchronic and chronic inhalation toxicity studies with experimental animals, as summarized in ATSDR

(2000). The TCA is substantially lower than all reported NOAELs and LOAELs from these studies. There is a precedent for use of this TCA in Canada. Within the Province of Ontario, the OMOE (2011) adopted the RIVM TCA as the inhalation TRV for non-cancer effects of PCBs. Overall, reliable human inhalation toxicity data for PCB mixtures remains limited within the scientific literature.

Table 8-4 presents two different oral TRVs for PCBs. The oral TRV selected for use in the HHRA was 0.02 µg/kg body weight/day (ATSDR, 2000; U.S. EPA, 1996; WHO, 2003). This value was selected over the Health Canada (2010b) TRV as it appears to have a stronger scientific basis and greater degree of consensus across regulatory agencies. It is also a more conservative value as it is nearly an order of magnitude lower than the Health Canada oral TRV. ATSDR (2000) reports the TRV of 0.02 µg/kg body weight/day was affirmed by a panel of experts (the panel included senior scientists from ATSDR, the U.S. Food and Drug Administration, U.S. EPA, the U.S. Centers for Disease Control and Prevention, the U.S. National Institute of Health, the U.S. National Institute for Occupational Safety and Health, Health Canada, various academic institutions, and toxicological research organizations/laboratories). ATSDR (2000) further notes that 0.02 µg/kg body weight/day is supported by available human and experimental animal data for neurodevelopmental effects. While this value is not based on such effects, it is protective of them, as neurodevelopmental effects have been reported to occur at higher doses than 0.02 µg/kg body weight/day in both humans and experimental animals. Furthermore, as the selected TRV is based on studies with Aroclor 1254, it is considered representative of the PCBs detected in soil at the PCB Area. Laboratory resemblance data for the PCB Area soil samples suggest Aroclor 1254 was among the more common PCB mixtures that were used within the transformers placed in this area (Aroclor 1242 and 1260 resemblance was also noted for some PCB Area soil samples).

The TRVs presented in Table 8-4 are based entirely on non-carcinogenic effects. It is acknowledged though that some jurisdictions (including the U.S. EPA and California EPA) consider PCBs to be human carcinogens. In addition, a recent re-evaluation of PCBs by the International Agency for Research on Cancer (IARC, 2013, *in press*) has updated the cancer classification for PCBs to Group 1 – carcinogenic to humans. In other words, IARC now considers PCBs as confirmed human carcinogens, which is a change from the previous IARC classification of probable human carcinogens for PCBs. Further details on the new IARC evaluation is available in a news release posted in The Lancet, Volume 14, April 2013 and in the soon to be released full IARC evaluation document (i.e., IARC, 2013, *in press*).

While it is not yet known how the new IARC evaluation may alter the existing TRVs for PCBs developed by regulatory agencies, there are presently a number of jurisdictions around the world that either do not classify PCBs as human carcinogens or have not developed cancer-based TRVs (such as slope factors or unit risk values) at this time, including Health Canada. Presently, according to information reported by Health Canada (2010c), PCBs are considered to be threshold carcinogens that act via a non-genotoxic mode of action, and as such, would not necessarily require TRVs that are expressed as slope factors or unit risk values. Although, the current Health Canada (2010b) TRV is not based on carcinogenic effects at

all, as indicated in Table 8-4. The human carcinogenicity of PCBs has been debated for decades within the scientific community and available human and experimental animal studies have been considered inconclusive by numerous regulatory agencies with no clear and consistent exposure or dose-response relationships (for such endpoints as tumour incidence, cancer mortality) having been established to date for either individual PCB congeners, groups of congeners, or commercial PCB mixtures, such as Aroclor formulations (WHO, 2003). Furthermore, in experimental animal studies, the doses associated with cancer have generally been much higher than those that caused other effects, including those that the non-cancer oral TRVs are based upon (WHO, 2003).

When there is scientific debate or controversy regarding a substances' human carcinogenicity status, or even when there appears to be scientific consensus that a certain substance is a human carcinogen, it does not necessarily mean cancer-based TRVs will be available for use in HHRAs. When any regulatory agency derives a human health-based TRV based on cancer effects, there are many factors that must be considered that relate to the overall weight of evidence for carcinogenicity, the consistency and nature of the reported exposure/dose response profiles, the consistency between different studies for cancer outcomes (i.e., same types of cancer? same tissues or organs affected?) and data quality and limitations/uncertainties associated with the key studies. It can often take a few to several years to develop and approve a cancer-based TRV. Thus, it is not uncommon for a given jurisdiction to have only non-cancer TRVs available for a substance, even when there is strong evidence to suggest that the substance is a human carcinogen.

While the current HHRA uses non-cancer based TRVs (which are believed to be the most appropriate to use at this time), the potential carcinogenicity of PCBs is considered further in the discussion of the HHRA results (Section 8.4).

8.3.3 Bioavailability/Bioaccessiblity and Route Extrapolation Considerations

Bioavailability and Bioaccessibility

The response of the body to chemical exposure depends on the quantity of the chemicals that actually enter the target organs, tissues and cells. In many cases, only a fraction of the chemicals ingested, inhaled or in contact with the skin are actually absorbed into the body.

Bioavailability refers to the extent and rate to which a chemical can be absorbed into the systemic circulation of an organism and potentially produce an adverse effect (Hrudey et al., 1996; Kelly et al., 2002). For HHRA, it is important to distinguish between two types of bioavailability: i) absolute bioavailability and ii) relative bioavailability.

Absolute bioavailability refers to the fraction or percentage of a compound which is ingested, inhaled or applied on the skin surface that is actually absorbed and reaches the systemic circulation (Hrudey et al., 1996). Absolute bioavailability is often defined as the ratio of an absorbed dose to an administered dose. In studies investigating absolute bioavailability, the absorbed dose is typically determined by measuring the concentration of the compound in blood over time or by measuring the mass of the compound in excreta (e.g., urine, feces or exhaled air). The absorbed (or internal) dose is useful for characterizing risk if the toxicity values describing the dose-response relationship (i.e., RfD, TDI, etc.) are based on the absorbed dose. However, most toxicity values are based on a delivered or administered dose rather than an absorbed dose; thus, in HHRA, it is usually not necessary to determine the absolute bioavailability of a contaminant (Schoof, 2003).

Relative bioavailability (RBA) is defined as a measure of the differences in extent of absorption between two or more forms of the same chemical (e.g., lead sulphide versus lead acetate), different exposure vehicles (e.g., food, soil and/or water), or different doses (Schoof, 2003; Kelly et al., 2002). Relative bioavailability is particularly important for HHRAs as matrix effects in some environmental media can substantially decrease the bioavailability of a soil- or sediment-bound chemical, relative to the soluble forms and dosing media that are generally used in the toxicology studies upon which TRVs are derived from (i.e., food, water). Thus, relative bioavailability is calculated as the ratio of the absorbed fraction from the exposure medium in the risk assessment (e.g., soil) to the absorbed fraction from the dosing medium used in the critical toxicity study. When relative bioavailability is expressed in this manner, it is commonly termed a RAF. Incorporating RAFs in HHRAs can greatly improve estimates of the external (i.e., administered) dose that is available for absorption (Schoof, 2003; U.S. EPA, 2007). It is important to recognize that while absolute bioavailability can never exceed 100%, relative bioavailability can exceed this value (U.S. EPA, 2007).

The bioavailability of a chemical is dependent on the chemical form, the environmental medium, as well as the tissues/organs with which the chemical interacts. Thus, when applying TRVs, it is important to consider the bioavailability of each chemical in the particular study from which the TRV is derived, to obtain reasonable estimates of the actual quantity of the chemical potentially entering the body.

A particularly important determinant of the oral bioavailability of chemicals present in soil is the soil bioaccessibility. The term “bioaccessibility” refers to the fraction of the substance that can be biologically extracted from the exposure media (i.e., soil in this case) and solubilized within the gastrointestinal tract so that it is available for absorption through the intestinal wall into the blood stream. In other words, bioaccessibility sets an upper limit on oral bioavailability and the two processes are positively correlated. In HHRAs that evaluate the soil ingestion route of exposure, bioavailability inherently includes bioaccessibility (Kelly et al., 2002). Thus, measures of bioaccessibility are used to estimate relative bioavailability. A number of recent studies and reviews suggest the degree of a substance’s bioaccessibility in soil depends greatly on both the soil and the contaminant properties (e.g., Oomen et al.,

2006; 2002; Schoof, 2003; Gron and Andersen, 2003; EAUK, 2005; Casteel et al., 2006; Wragg et al., 2007; Palumbo-Roe and Klinck, 2007).

The use of bioavailability and bioaccessibility information in HHRA is rapidly gaining increased attention by both risk assessment practitioners and government regulators, as it directly impacts human health risk estimates (for the soil ingestion pathway) and the development of site-specific soil remedial objectives. Including soil bioaccessibility information as part of the HHRA process can allow for more realistic estimates of the systemic exposure to chemicals from soil and dust ingestion, relative to using generic default assumptions that are frequently excessively conservative and unrealistic (EAUK, 2005).

There are two main ways in which the RBA of chemicals in soil are estimated. Traditionally, *in vivo* studies (i.e., animal studies) have been used to determine RBA directly; however, these studies can have significant time and cost constraints associated with them (Ruby et al., 1999) and are rarely conducted. A much more widely used alternative is the use of *in vitro* extraction studies that simulate the human gastrointestinal tract in order to estimate the mobilization of compounds from soil during the digestion process. These *in vitro* tests offer a rapid and inexpensive means of estimating soil bioaccessibility and can provide a reasonable, yet conservative approximation of RBA. *In vitro* soil bioaccessibility tests can be conducted using a gastric phase only or with both a gastric and intestinal phase. It must be recognized though that bioaccessibility does not equal RBA *per se*; however, bioaccessibility may be considered a reasonable approximation of RBA for substances lacking adequate *in vivo* RBA data.

Selection of Oral, Inhalation and Dermal RAFs for the HHRA

With respect to the current HHRA, site-specific data on the *in vitro* soil bioaccessibility of PCBs has not been collected at this time. Furthermore, the scientific literature regarding the *in vitro* soil bioaccessibility of PCBs is limited and variable, and the available data on the *in vivo* oral RBA of PCBs is equally or more limited and variable. Furthermore, there is not yet consensus on what the most appropriate *in vitro* soil bioaccessibility test protocols would be for PCBs in soil, and the soil properties that control or influence PCB soil bioaccessibility (with respect to human gastrointestinal physiology and conditions) are not well characterized at this time. Thus, the HHRA assumed an oral RAF of 1.0 for the soil ingestion exposure pathway. This is a common conservative assumption for organic chemicals in HHRAs, and within this HHRA, is equivalent to assuming the oral bioavailability of PCBs in soil is the same as the oral bioavailability of these substances in the toxicity studies from which the oral TRVs were developed. OMOE (2011) also currently uses 1.0 as the default oral RAF for PCBs for the soil ingestion pathway.

The HHRA assumed inhalation exposures to PCBs were 100% bioavailable (RAF = 1.0) or as bioavailable as in the studies that the inhalation toxicity reference values were derived from. This is consistent with Health Canada (2010a) guidance.

Dermal RAFs used in the HHRA are described below, in the context of oral to dermal route extrapolation.

Route-to-Route Extrapolation

It is necessary to consider route-to-route extrapolation when a TRV is not available for the exposure route of interest and no other data (such as pharmacokinetics) are available. For example, it is common in HHRA to assess the potential risks posed by dermal exposure based on TRVs that are established for the oral exposure route. In doing so, the systemic dose that is absorbed dermally is scaled to the 'equivalent' oral dose by adjusting for the bioavailability of the dermally-applied chemical relative to an orally-administered dose (i.e., relative bioavailability adjustment).

Other situations where adjustments for bioavailability may be made in an HHRA are as follows, where sufficient data exist:

- When the effects of the chemical of interest are systemic in nature (i.e., following entry into and distribution by the bloodstream, as opposed to effects occurring only at the site of entry [e.g., lungs, skin, gut]);
- When the medium of administration or medium of exposure results in different relative bioavailabilities (e.g., ingestion in drinking water versus ingestion in soil); and,
- If the bioavailability of the chemical, based on the particular study animal/receptor, is different from that of the assessment receptor (e.g., the published TRV is based on a study using rats, the receptor of interest is a human, and there are reported different bioavailabilities for the chemical between species).

When oral-to-dermal route extrapolation is necessary, the relative absorption or bioavailability difference between the oral and dermal routes of exposure can be expressed as a RAF (i.e., RAF_{dermal}). This RAF, calculated as follows, is applied to dermal exposure estimates to adjust these exposures prior to their comparison with oral TRVs.

$$RAF_{dermal} = \frac{AF_{dermal}}{AF_{oral}}$$

Where:

RAF_{dermal}	=	the dermal relative absorption factor.
AF_{dermal}	=	the fraction of the applied chemical absorbed through the skin.
AF_{oral}	=	the fraction of the ingested chemical absorbed via the gastrointestinal tract.

For PCBs, a dermal RAF of 0.14 is currently recommended by both Health Canada (2010a) and OMOE (2011). This RAF was applied to the dermal exposure estimates in the HHRA[2].

8.3.4 Risk Characterization

Risk characterization is the final step in an HHRA and integrates the exposure and hazard (toxicity) assessments to provide an estimate of human health risk for the receptors, COPCs, exposure pathways and exposure scenarios that were evaluated. Potential risk is characterized through a comparison of the estimated or predicted COPC exposures to the selected human receptors from all exposure pathways and routes (from the Exposure Assessment) with the identified TRVs from the Hazard (Toxicity) Assessment.

For chemicals with TRVs based on threshold effects (non-carcinogens), this comparison takes the form of a ratio calculation, and is referred to as the HQ, which is also sometimes referred to as an exposure ratio (ER) or risk quotient (RQ). The HQ is calculated by dividing the estimated level of exposure by the TRV, as indicated in the following equation:

$$\textbf{Hazard Quotient} = \frac{\textbf{Estimated Exposure (µs/kg body weight/day)}}{\textbf{TRV (µR/kg body weight/day)}}$$

For chemicals with TRVs based on non-threshold dose response relationships (i.e., carcinogens), risk characterization typically involves determining an ILCR level that would be expected given the predicted exposures. The ILCR is calculated by multiplying the predicted exposure by the slope factor (or unit risk value) for the COPC in question[3]. The ILCR is defined as the predicted risk of an individual in a population of a given size developing cancer over a lifetime, and is expressed as the prediction that 1 extra person per n people, exposed at the stated rate daily over a lifetime, would develop cancer, where the magnitude of n reflects the risks to that population – the larger n is, the smaller the risk (e.g., if the ILCR is 1 person per 10, the predicted risks of any individual developing cancer would be higher than if the ILCR is 1 per 1,000). Calculated ILCRs are then compared to "acceptable risks" or "target cancer risk levels" to evaluate the significance of the ILCR. In Newfoundland and Labrador, and at federally owned sites in Atlantic Canada, an ILCR (facility or site-related) of one-in-one hundred thousand (i.e., one additional cancer per hundred thousand people) is considered to be acceptable.

The following equation provides the method by which the ILCR is calculated:

[2] The dermal RAF is the same as the AF terms provided in the dermal exposure equations presented previously in Section 4.3.1.

[3] Alternatively, the risk characterization of carcinogens can be conducted in the same manner as the risk characterization of non-carcinogens if the cancer-based slope factors or unit risks are converted to risk-specific doses or concentrations that correspond to the acceptable target cancer risk level.

ILCR = Estimated Exposure (µL/kg body weight/day) x CSF ([µk/kg body weight/day]$^{-1}$)

Where:

ILCR = Incremental Lifetime Cancer Risk
CSF = Cancer Slope Factor

The resulting estimated cancer risk (ILCR) can then be compared to the acceptable target cancer risk level (i.e., 1 in 100,000) to determine if estimated COPC exposures pose an unacceptable human health risk.

As PCBs are assumed to be non-carcinogenic to humans in this HHRA (See discussion in Section 8.3.2), only HQ values are estimated.

It is important to recognize that HQ and ILCR values are not absolute measures of risk, nor are they measures of actual risk; rather, they are most appropriately considered as indicators of potential human health risks which enable the following:

- Compare potential adverse health effects between COPCs and between different exposure scenarios (e.g., different site-specific conditions, different site use options);
- Estimate potential adverse health effects from exposures to mixtures of COPCs that may elicit similar effects in organs, tissues or cells (e.g., all chemicals that cause liver toxicity, kidney toxicity, respiratory tract cancers or respiratory irritation effects); and,
- Simplify HHRA results to provide a clear understanding of the results and an appreciation of their significance.

All HQ values are presented in the HHRA results section (i.e., Section 8.4) as point estimate values.

Another key aspect of risk characterization is to identify and consider uncertainty within the HHRA. A discussion of key areas of uncertainty in the HHRA is presented in Section 8.5.

8.3.4.1 Interpretation of HQs and ILCRs

Once HQ values have been determined for threshold (non-carcinogenic) chemicals, they are then compared to a target HQ that is essentially an indicator of "safety". In general, if the total chemical exposure from all relevant exposure pathways is equal to or less than the TRV, then the HQ would be 1.0 or less, and no adverse health effects would be expected. In this case, the target HQ would be 1.0, assuming there are estimates of exposure from all relevant exposure pathways.

However, in Canadian HHRAs where a number of potential human exposure pathways are excluded from evaluation (due to limited data, uncharacterized exposure sources, etc.), which is very common for

contaminated site HHRAs, the target HQ cannot be 1.0, as a number of pathways and routes would not be considered. Rather, a lower target HQ would be selected. The reason for this is non-carcinogenic (threshold chemicals) TRVs represent the level of total exposure that would not result in adverse health effects, regardless of the sources or pathways of exposure. If a HHRA evaluates only single sources of contamination and/or a limited number of exposure pathways, selecting a target HQ of 1.0 may not be appropriate or adequately protective. In many HHRAs conducted across Canada, a default target HQ of 0.2 is typically used when the assessment is only investigating exposure from selected pathways, sources and routes. This implies a maximum of 20% of the total exposure has been apportioned to any one source or pathway and is based on simple subdivision of the five main categories of human chemical exposure (i.e., intake from: air, water, soil/sediments, and diet and consumer products). A target HQ of 0.2 is endorsed and/or recommended by various regulatory agencies in Canada including Health Canada (2010a), OMOE (2011), and CCME (2006) when limited exposure pathways, sources and routes are assessed. This has also become common practice in HHRAs conducted under the Atlantic Canada RBCA process in recent years. Thus, in this HHRA, the target HQ is set at 0.2.

For COPCs that are non-threshold chemicals (i.e., chemicals believed to act as carcinogens), the calculated ILCR is compared to the accepted target cancer risk level (i.e., 1 in 100,000). A predicted ILCR of less than 1 in 100,000 (<0.00001) is considered acceptable within Atlantic Canada, while a predicted ILCR of equal to or greater than 1 in 100,000 is considered to be a potentially unacceptable level of risk. The risk characterization of carcinogens is based on 100% of the acceptable target risk level of 1 in 100,000 (i.e., there is no apportioning of cancer risks), as ILCR values are independent of background exposure sources; Health Canada, 2010a).

If non-cancer risk estimates (i.e., HQ values) are less than 0.2 in an HHRA, no adverse health effects would be expected to occur given the COPCs, receptors, exposure pathways and scenarios evaluated. As HHRAs typically use a number of conservative factors and assumptions, there is usually a high degree of confidence that such HQ values do not pose a potential human health risk. If HQ values are greater than 0.2, there may be a potential for adverse effects in sensitive individuals or in one or more of the exposure scenarios considered. However, given the conservatism typically employed in HHRAs, calculated HQ values greater than the target HQ value do not necessarily indicate adverse health effects are likely to occur. Generally, in cases where HQ values are greater than 0.2 in an HHRA, assumptions, parameters and data used within the HHRA are closely re-examined prior to concluding whether or not a health risk exists.

Similar to the interpretation of HQ values, ILCR values less than 1 in 100,000 suggest adverse health effects would not be expected to occur given the COPCs, receptors, exposure pathways and scenarios evaluated. However, ILCR values greater than 1 in 100,000 do not necessarily indicate a potential cancer risk exists. Rather, when ILCRs exceed the acceptable target cancer risk level, assumptions, parameters

and data used in the HHRA are typically re-examined prior to making final conclusions regarding the likelihood of adverse health effects occurring.

For COPCs that have different effects, target organs and/or mechanisms of action by the oral and inhalation routes, it is standard HHRA practice that exposures for all oral and dermal pathways be summed separately from all inhalation pathways and then compared separately to the pathway-specific TRVs, with the resulting risk estimates compared separately to the target HQ or ILCR value. For such COPCs, there may be different TRVs for the oral and inhalation routes that reflect differences in target tissues, endpoints, effects and mechanisms of action. This is not conducted for COPCs where the same or similar effects are reported to occur *via* both the oral and inhalation routes or for COPCs where only an oral TRV exists. Rather, in these situations, oral, dermal and inhalation exposures or risks can be summed together.

For PCBs, the basis of the inhalation TRV (i.e., RIVM TCA from Baars et al., 2001) is not well documented. Thus, it is assumed the health effects of PCBs via inhalation are the same or similar to those that occur via the oral route of exposure.

Other considerations that can be important when interpreting HQ and ILCR values include: considering other potential sources of COPC exposure; the impact of uncertainty, variability, data gaps; and the assumptions made in the Problem Formulation, Exposure Assessment and Hazard (Toxicity) Assessment steps, on the magnitude of the human health risk estimates.

8.3.4.2 Consideration of Chemical Mixtures and Potential Toxicological Interactions

While PCBs are the only chemicals evaluated in the HHRA, it is important to acknowledge people accessing the Site would incur potential exposures to substances other than PCBs at the same time they would incur exposures to PCBs. The following paragraphs briefly discuss the concepts of exposure to chemical mixtures and the potential toxicological interactions that may occur.

Most HHRAs evaluate health risks related to COPCs on an individual basis. However, humans are typically exposed to complex mixtures of substances, rather than to single substances. Concurrent exposures to multiple chemicals may result in toxicological interactions of these chemicals at target sites in the body. These interactions may result in a combined toxicity that are equal to the sum of the toxicities of the individual chemicals (additivity), greater than the sum (synergism or potentiation) or less than the sum (antagonism). For example:

- additivity (1 + 1 = 2)
- antagonism, inhibition (1 + 1 = 0)
- synergism (1 + 1 = 3)
- potentiation (1 + 0 = 2)

Definitions for the specific types of interactions that may occur are as follows (ATSDR, 2004a):

- Additivity: when the effect of the mixture can be estimated from the sum of the exposure levels (weighted for potency) or the effects of the individual components.
- No apparent influence: when a component which is not toxic to a particular organ system does not influence the toxicity of a second component on that organ system.
- Synergism: when the effect of the mixture is greater than additive on the basis of the toxicities of the components.
- Potentiation: when a component that does not have a toxic effect on an organ system increases the effect of a second chemical on that organ system.
- Antagonism: when the effect of the mixture is less than additive on the basis of the toxicities of the components.
- Inhibition: when a component that does not have a toxic effect on a certain organ system decreases the apparent effect of a second chemical on that organ system.
- Masking: when the components produce opposite or functionally competing effects on the same organ system and diminish the effects of each other, or one overrides the effect of the other.

It is important to recognize the likelihood of a biologically significant interaction occurring is a function of at least the physical, chemical and biological properties of the chemicals involved, their modes of toxic action and their environmental media concentrations. Most greater than additive interactions can only be demonstrated at high exposure rates, where clear adverse effects are observed. Such interactions have not been observed or quantified at the relatively low rates of exposure typical of those associated with most environmental or occupational situations (NAS, 1983; Krewski and Thomas, 1992) and are therefore not typically considered in risk assessments. Additivity is generally recognized as the most plausible type of interaction that may occur in situations of chemical exposure in the ambient environment. However, it requires the chemicals act through the same or similar mechanisms of action and/or affect the same target organs or tissue(s). In HHRAs where the COPCs are known to act via different mechanisms of toxic action and affect different target organs or tissues, it is typically assumed that no potential toxicological interactions warrant consideration and the estimated exposures and risks for the COPCs are considered separately.

Complicating the assessment of toxicological interactions is the fact that most health criteria, guidelines, TRVs and other health-based benchmarks are derived for individual substances or mixtures of similar substances and do not account at all for concurrent exposures to other substances.

At this time, regulatory agencies typically recommend HHRAs evaluate the individual substances that have been identified as COPCs and then determine whether or not the exposures or risks for the individual COPCs in the mixture could reasonably be considered additive, based on the health effects associated with each substance. In this HHRA, because data are available only for total PCBs, these

substances are inherently assessed as a mixture. The selected TRVs are representative of the PCB mixtures (i.e., Aroclor 1254, 1260, 1242) that have been identified in soil.

While information on toxicological interactions of PCBs with other substances is available (e.g., ATSDR, 2004b,c,d; 2000), consideration of such interactions and their potential implications with respect to human exposures and risks is not within the scope of this HHRA, which focuses on PCBs only. However, because of the diversity of biological responses that can be induced by PCB mixtures, there is believed to be a high potential for PCB mixtures to alter the toxicity of other chemicals or for other chemicals to alter the toxicity of PCBs, particularly with respect to other chlorinated organic substances, many of which have common metabolic pathways in humans and similar modes of toxic action (ATSDR, 2000). In this context, "alter" does not necessarily mean increase, as there is evidence that PCB mixtures can reduce the toxicity of some other substances (see ATSDR, 2004b,c,d; 2000). Despite this high likelihood for interactions, data limitations within the toxicological literature would be expected to preclude quantitative assessment of toxicological interactions involving PCBs, for the most part.

8.4 Human Health Risk Assessment Results and Conclusions

8.4.1 Qualitative Assessment of Potential Soil to Outdoor Air Migration of PCBs

As previously mentioned, it is possible that trace amounts of PCBs could occur in outdoor air as vapours as a result of volatilize from soils within the PCB Area. As such, the outdoor air inhalation pathway was considered in the HHRA. This exposure pathway is assessed qualitatively by comparing soil concentration data (from all depths - subsurface and surface) to the applicable OMOE (2011) soil to outdoor air (S-OA) component value for coarse-grained soil, non-potable groundwater use and commercial land use (i.e., 120 mg/kg).

The OMOE (2011) S-OA values are calculated using the Jury Reduced Solution Finite Source Volatilization Model and a number of conservative assumptions that are based on environmental conditions within Ontario. The S-OA value for PCBs is also based in part on the Baars et al., (2001) TCA of 0.5 µg/m^3, which is the same inhalation TRV used in this HHRA. Essentially, the S-OA value for PCBs is the soil concentration that will not result in an exceedance of the TCA if conditions assumed in the Jury model are met, and PCBs are able to volatilize from soil. While there is some uncertainty in the application of these S-OA values to the Site (i.e., no local wind or atmospheric mixing data is available), they represent a conservative set of benchmarks towards determining if PCB concentrations in soil are present at concentrations that could pose a potential concern with respect to the outdoor air inhalation pathway.

Table 8-5 compares the summary statistics of the PCB concentrations in soil to OMOE (2011) S-OA value.

Table 8-5 Comparison of Soil PCB Concentrations to the OMOE (2011) S-OA Soil Component Value

PCB Soil Concentration Statistics		OMOE (2011) S-OA Value for PCBs; Commercial Land Use, Coarse-grained Soil; Non-potable Groundwater Condition (mg/kg)
Statistic	**Values**	
Maximum	600 mg/kg	120
Minimum	0.043 mg/kg	
Arithmetic Mean	22 mg/kg	
Median	1.5 mg/kg	
UCLM95	51[a] mg/kg	
95th Percentile	88 mg/kg	
Frequency of Exceedance of Soil PCB Concentrations over S-OA value	4 samples (3.2%)	

Notes:
N=127 (subsurface and surface soil PCB data for PCB Area; for soil samples representative of locations where soils were not excavated and removed; as PCBs are known to be persistent and tend to degrade slowly in soils, all representative soil data from 2003 to 2013 were considered).
For soil samples with a laboratory or field duplicate sample, the higher result of either the original sample or its duplicate was retained for HHRA purposes.
All <RDL values were set equal to RDL that was achieved, for the purposes of calculating summary statistics.
a 95% Chebyshev (Mean, Sd) UCLM95 calculated with ProUCL v4.1.

As presented in Table 8-5, few soil samples had PCB concentrations exceed the OMOE (2011) S-OA value. The summary statistics that represent central tendency (i.e., arithmetic mean, median), upper bound of central tendency (i.e., UCLM95) and upper bound (i.e., 95th percentile) are below the S-OA value. Only the maximum soil PCB concentration (and three other samples) exceeds the S-OA value. This results in a frequency of exceedance over the S-OA value of 4 samples (3.2%). In HHRA, it is common to use a "cutoff value" of 5% for determining the significance of the frequency of exceedance of soil data over benchmark values. A significance level of 5%[4] is consistent with the statistical significance level that is commonly used in many statistical tests for environmental applications. The frequency of exceedance of the PCB concentrations in soil over the S-OA value is less than 5%.

Given these considerations, PCB concentrations in soil are not considered to pose a human health concern for the soil to outdoor air exposure pathway. This conclusion is supported when considering the uncertainty inherent in estimating the volatilization potential of PCBs. As previously noted in Section

[4] While there is no consistent regulatory guidance on a cut-off value to use when considering the frequency of detection, or frequency of exceedance over benchmarks or reference concentration statistics, a value of 5% is often used in practice as it is consistent with the significance level used in typical statistical comparison testing. Also, within the Superfund program, the U.S. EPA has used a 5% cut-off value to eliminate chemicals from further consideration since the early 1990s. U.S. Navy (2008) HHRA guidance also supports the use of a 5% cut-off value when considering detection and benchmark exceedance frequency, but cautions that one must consider the sample size when choosing a cut-off value (as 5% is not appropriate if one has a sample size of <20; in these cases, professional judgement is used to determine if frequency considerations are appropriate and what an alternate cut-off value could be). The U.S. Navy also cautions that one must consider the spatial distribution of the chemicals of interest, when deciding whether or not to eliminate chemicals (or exposure pathways) from consideration on the basis of detection or benchmark exceedance frequency.

8.2.1, the potential for volatilization of PCBs from soil to air is dependent on a number of conditions including the depth at which PCBs occur, the type of PCB mixture and its congener profile (for example, lower chlorinated congeners tend to have a higher vapour pressure than higher chlorinated congeners; the higher chlorinated congeners also tend to sorb more tightly to organic matter in soils and sediments), the age of the PCB mixture (as most organics have a decreasing potential for volatilization with increasing time in a soil matrix due to sorption, weathering, biodegradation and sequestration effects), and various soil properties such as texture, organic carbon content, pH, temperature, moisture content, aeration/oxygenation, presence of microbial communities that can biodegrade or sequester PCBs and the degree of ground vegetation cover. The Jury model does not account for these factors that can limit or otherwise influence volatilization potential.

Current Scenario (As Is - Outdoor Site Visitor (Trespasser))

Results of the quantitative HHRA for the Current Scenario (As Is - Outdoor Site Visitor (Trespasser)) are presented in Table 8-6.

Table 8-6 Human Health Risk Estimates

COPCs	UCLM95 Site Soil Concentration (EPC); mg/kg	Female Toddler Receptor HQ	Benchmarks of Safety (Target HQ and ILCR Values)
Oral and dermal soil exposure pathways			
PCBs	27	0.71	HQ of 0.2
Inhalation soil exposure pathway			
PCBs	27	0.0000039	HQ of 0.2

Notes:
HQ values and soil concentrations are rounded to two significant figures.
Shaded cells indicate exposure estimates exceed the benchmark of safety.
Risk estimates are presented separately for oral + dermal and inhalation pathways as PCBs have different TRVs available for the oral and inhalation routes of exposure.
Risk estimates presented are based on the upper 95% confidence limit of the arithmetic mean surface soil (≤1.5 m depth) concentration (i.e., 95% Chebyshev (Mean, Sd) UCLM95 calculated with ProUCL v4.1); N = 64. The UCLM95 was calculated for surface soil samples that are representative of locations where soils were not excavated and removed. As PCBs are known to be persistent and tend to degrade slowly in soils, representative soil data from 2003 to 2013 were considered. In calculating the UCLM95, the highest result of either an original sample or its duplicate (field and/or lab) was retained for HHRA purposes, and <RDL values were set equal to the RDL that was achieved for a given soil sample.

As presented in Table 8-6, the inhalation risk estimates for PCBs do not exceed the target HQ of 0.2 when exposures are based on UCLM95 surface soil concentrations. These risk estimates suggest no adverse health effects would be expected to occur in female toddler receptors exposed to PCBs in soil via the inhalation route of exposure. Given the conservative assumptions used within the HHRA, and since female toddlers tend to receive greater exposures than males and other human receptor classes on a body

weight basis, no adverse health effects would be expected to occur in other human receptors with respect to the inhalation route of exposure.

For the combined oral and dermal routes of exposure, the exposure estimates exceed the target HQ of 0.2 when exposures are based on UCLM95 surface soil concentrations. As the degree of exceedance over the target HQ is not large (and considering the conservative assumptions used within the HHRA), these risk estimates suggest an overall low potential for adverse health effects in female toddler receptors exposed via the oral and dermal exposure routes to PCBs in soil. Other human receptor classes (and male receptors in general) would incur lower estimated exposures and would have a lower potential for risk, relative to the female toddler receptor.

As previously noted, it is considered toxicologically appropriate to sum risk estimates across COPCs and/or across oral + dermal and inhalation exposure routes, only if the target organs and effects (including modes of toxic action) are the same or similar between COPCs, and/or between these exposure routes for a given COPC. This is consistent with Health Canada (2010a) guidance. Although the basis of the inhalation TRV for PCBs is not well documented (discussed in Section 8.3.2), it can be assumed that similar effects would occur following inhalation, oral or dermal exposures to PCBs. Review of the available inhalation toxicology data for PCBs relative to the data obtained from oral or dermal exposure studies suggests this is a reasonable assumption. Thus, it was considered appropriate to sum the oral, dermal and inhalation risk estimates for PCBs. The resulting combined risk estimate does not increase significantly when inhalation risk estimates are added to the oral + dermal risk estimates.

As previously noted in Section 8.3.2, there is some uncertainty and debate over the carcinogenicity of PCBs to humans. While there now appears to be consensus in most jurisdictions that PCBs are human carcinogens (i.e., IARC, 2013, *in press*), cancer-based TRVs have not been developed by Health Canada to date. It has been Health Canada's position in the recent past that the evidence for human carcinogenicity of PCBs is inconclusive. It is not yet known if or how the recent IARC reclassification of PCBs as essentially confirmed human carcinogens will change Health Canada's position, and/or result in adoption or derivation of cancer-based TRVs. Given the current carcinogenic status of PCBs in Canada, this HHRA assesses these substances as non-carcinogens.

However, recognizing PCBs are potentially human carcinogens, this HHRA also examines the impact of assuming PCBs are carcinogenic on the human health risk estimates. If the currently available and most conservative cancer-based TRVs for PCBs from the U.S. EPA were to be used in the HHRA, risk estimates for the oral + dermal routes of exposure would increase by approximately an order of magnitude, which would suggest a relatively high potential for risk for the receptors and scenarios evaluated in this HHRA. Similarly, human health risk estimates for the inhalation route of exposure would increase by approximately 5-fold, but remain below levels associated with a potential for adverse human health effects.

Given these outcomes, the uncertainty related to the current carcinogenic status of PCBs in Canada and the exceedance of the target HQ (assuming PCBs are not carcinogenic to humans) for oral and dermal exposure estimates (Table 8-6), the need to consider corrective action in relation to soil PCB concentrations at the Site are warranted, despite the high likelihood that the conservative assumptions used within the HHRA have substantially overestimated potential exposures and risks. This is in keeping with common site management practices for PCBs (where some corrective action is often taken irrespective of human health and/or ecological risk potential), and is also in keeping with the fact that PCBs are frequently associated with a high potential for public concern.

While assuming PCBs are human carcinogens leads to higher human health risk estimates, it should be recognized that concerns regarding cancer development in HHRAs are primarily associated with situations of chronic or continuous exposure to a given carcinogen(s), where there is a high frequency and duration of exposure to carcinogenic substances. This is not the case for the Site where human exposure to PCB-impacted soils would occur on a transient or intermittent basis and for short durations when it does occur. Such exposure conditions are generally not associated with a high potential for developing cancer. There are no known studies to date (for any carcinogen) that have been able to demonstrate a strong link between cancer development and short term, low frequency, low duration exposures to the generally low levels of carcinogenic chemicals that may occur in ambient environmental media.

With such conclusions as those reached above, an HHRA would typically develop soil SSTLs (i.e., SSTLs are essentially maximum possible soil concentrations of COPCs that would not pose a potential risk of adverse health effects, or in other words, exceed the target HQ and/or ILCR values). SSTLs are often back-calculated or iterated from a site-specific HHRA model. However, given the current uncertainty regarding the carcinogenic status of PCBs in Canada, soil SSTLs for the Site could vary by an order of magnitude or more depending on whether or not PCBs are assumed to be human carcinogens. Furthermore, and most importantly, the PCB Area will be covered with an engineered cap as part of the closure activities. Because of this, there is considered to be no merit in developing soil SSTLs, when it is known that potential human exposures and risks to PCB-impacted soils will be mitigated by the engineered cap. The implications of this cap on potential human exposures and risks are evaluated qualitatively within the Future Scenario (Post-Capping), which is discussed below.

8.4.2 Future Scenario (Post-Capping)

Because the closure plan for the WDS will include the placing of an engineered cap over the PCB Area, a secondary HHRA exposure scenario was evaluated to determine if the cap is likely to minimize potential human exposures and risks. This Future Scenario (Post-Capping) is evaluated qualitatively based on such considerations as depth of the cap layers, the types of cap layers, the outcomes of the Current Scenario (As Is - Outdoor Site Visitor (Trespasser)) and review of the environmental media and biota chemistry data that has been collected to date in relation to the Site.

Although the final design of the engineered cap has not been completed, the cap will be approximately 2.0 m thick and consist of imported clean fill, bedding material, a geomembrane liner and a vegetative layer. The top layer will be graded to prevent ponding of water.

Thus, the engineered cap will result in a 2.0 m physical barrier that will prevent human contact with PCB-impacted soils. This will negate direct soil contact pathways for all residual PCB concentrations in what is currently surface and subsurface soil within the PCB Area. The only human exposure pathway that may remain operable once the cap is in place is soil to outdoor air migration of PCBs. However, as previously discussed (Section 8.4.1), this pathway is not considered to result in unacceptable potential human health risks that would warrant mitigation. Furthermore, the depth and types of materials that will comprise the engineered cap would be expected to substantially attenuate any residual PCBs that may volatilize and migrate to outdoor air through the soil profile and cap layers. It is considered likely that PCB vapour migration to outdoor air will be negligible once the cap is in place.

In conclusion, it is anticipated that the engineered cap will fully mitigate potential human exposures that were estimated in the Current Scenario (As Is - Outdoor Site Visitor (Trespasser)). If there is no or negligible exposure, then there is no or negligible potential for human health risks, irrespective of the current uncertainty regarding the carcinogenic status of PCBs in Canada. Furthermore, there is no evidence to date that PCBs are migrating from the PCB Area to off-site terrestrial or aquatic media or biota, based on the groundwater, surface water, sediment, and fish monitoring programs that have been conducted since 2003. With the exception of fish, these monitoring events have now occurred over multiple sampling events in multiple years and have consistently indicated PCB concentrations below the laboratory RDL. The engineered cap will further reduce the potential for PCBs to migrate off-site by removing potential surface media transport pathways such as wind erosion of impacted soil and bulk transport of impacted soil via surface runoff.

8.5 Uncertainty Analysis

In any risk assessment, the findings are based on available data from the specific study area or site and scientific literature, which are used in conjunction with a number of assumptions. Efforts were made to consider and apply appropriate assumptions and the existing data to adequately represent conditions at the Site. However, data is often limited, which inherently results in uncertainty in the assessment. Where uncertainty exists, assumptions are made and data is selected so as to err on the conservative side. The major sources of uncertainty, limitations and conservatism associated with this HHRA are presented below. Overall, given the tendency for the conservative assumptions used in the HHRA to overestimate PCB exposures, it is considered likely that the HHRA has substantially overestimated potential human health risks.

- In contaminated site HHRAs, it is typical to consider soil ingestion and dermal contact as "event driven". In terms of exposure frequency and duration values that are used in the HHRA for these exposure pathways, the event-driven assumption amounts to not considering hours per day on a site, but only considering days per week and weeks per year. Although, for developmental toxicants such as PCBs, the weeks/year term is also typically not considered, leaving only the days/week term to represent exposure frequency and duration. These are inherently conservative assumptions, as there is considerable uncertainty surrounding how soil contact exposure actually occur, and whether or not it is truly event-driven or more related to time spent on-site. In reality, it is probably a combination of both. For sites that have a land use that corresponds to a large amount of time spent on-site (such as agricultural, residential or commercial/industrial), the event-driven assumption is logical and appropriate, as the probability of a contact event occurring is related to the amount of time spent on a site. However, this assumption is less reasonable/realistic for sites where the land use is recreational or where site access by humans occurs in a manner that is transient, infrequent or intermittent (as is the case for this Site), as the probability of a contact event occurring is again related to the amount of time spent on a site. When exposure frequency and duration are amortized to reflect the event-driven assumption for direct soil contact exposure, the assumed time spent on-site for low access land use scenarios ends up being much greater than is realistic or plausible for the actual Site access situation being assessed. Thus, for low access sites, the event-driven assumption results in substantial overestimates of exposure and risk via the soil ingestion and dermal contact pathways and can lead to inappropriately identifying a need for corrective action, when there is no risk. This situation is further exacerbated when the COPCs being assessed are believed to be developmental toxicants. While there are HHRAs of sites with low/limited human access, where deviation from the event-driven assumption has been accepted through both peer and regulatory review processes, it is very rare for this to occur and be accepted in HHRAs conducted within Canada at this time.
- Treatment of data for the HHRA was conducted in a manner that is intentionally conservative (i.e., tending towards overestimation of exposure and risk). For example, concentrations of PCBs in soil below the laboratory RDL were assumed to be present at the RDL, and, for soil samples with field and/or laboratory duplicate samples, the higher concentrations of PCBs between the original and duplicate soil sample were retained for assessment.
- The HHRA assumes the entire outdoor dust level would be completely inhaled (i.e., all dust is comprised of the respirable $PM_{2.5}$ fraction). In reality, the outdoor dust level assumed in the HHRA is for total suspended particulate (TSP). Of this, it is expected that a much smaller proportion of dusts would be in the inhalable or respirable range.
- The use of deterministic (or point estimate) exposure analysis techniques in the HHRA is an approach that tends to overestimate potential exposures and risks. In deterministic exposure analysis, single concentrations representing reasonable maximum or upper bound exposure are typically used to represent the soil contaminant concentrations on the Site. This was the case in this HHRA, where UCLM95 soil concentrations of PCBs were evaluated as the EPCs.

- The use of the female toddler human receptor in the HHRA results in a conservative assessment that is protective of less-exposed and/or less sensitive receptors (i.e., male toddlers, older children, adolescents and adults). In addition, the toddler is considered an unlikely human receptor to access the Site.
- The HHRA assumed there is no attenuation, transformation, degradation or loss of PCBs in soil.
- The assumption that all airborne dusts which human receptors are exposed to while on the Site, come from the soils on the site, is conservative. Typically, the ambient dust level in a given area is attributable to a number of sources, including the Site under consideration, but also traffic, and various local off-site activities (such as landscaping, construction), as well as dust migration due to wind erosion of nearby fields, gravel roads or other areas with exposed soils.
- No alteration in human receptor exposure patterns due to inclement weather (e.g., rain, winds, very hot or very cold temperature) was assumed. It is likely that exposure to PCBs in soils would be limited during heavy rain or high wind events, as well as, on days of high heat and humidity or extreme cold. These meteorological factors would reduce the receptors' time outdoors on the Site, thereby reducing potential exposures to PCBs in soils. Not accounting for such factors likely overestimates potential exposures and risks. While a winter cover factor was applied to soil exposures, this factor does not address rain, wind or hot weather events that may also limit the time spent in contact with soils.
- Potential transfers of PCBs to the developing fetus and to breast milk in pregnant and/or lactating women (with subsequent exposure to nursing infants) was not evaluated in the HHRA. While there is abundant evidence that PCBs will cross the placental barrier, transfer to cord blood and also transfer to breast milk, the main pathway of concern in relation to elevated PCB levels in these biological fluids and tissues is chronic consumption of PCB-contaminated fish (ATSDR, 2000). Intermittent or transient exposure to PCB-impacted soil at a location with a small spatial area (as is the case for the Site) is much less likely to lead to elevated PCB levels in breast milk, cord blood and fetal tissues. While fish are caught and consumed by local community members, consumption does not typically occur throughout the year, and furthermore, all fish sampling to date has indicated non-detectable PCB concentrations from areas most likely to be influenced by the PCB Area of the WDS. In addition, the accumulation of PCBs in breast milk, cord blood and fetal tissues is known to be variable, influenced by a number of maternal and dietary factors, and is difficult to predict with accuracy (ATSDR, 2000). Lastly, given the PCB Area is located at a WDS, the likelihood that pregnant women or nursing mothers would spend sufficient time at the PCB Area such that they may incur PCB doses of sufficient magnitude and duration to lead to biologically significant transfers to the developing fetus and/or breast milk, is considered to be very low and is probably negligible.
- While the lack of detectable PCB concentrations in fish fillet samples (collected in 2006) suggests PCBs have not migrated to local water bodies, it must be recognized that fillet tissues were analyzed raw, with the skin and associated subcutaneous fat removed. This is not necessarily the most appropriate manner in which to prepare and analyze fish tissues when the concern is related to PCB exposure via fish consumption. It is known that PCBs accumulate preferentially in fatty tissues and

that PCBs can migrate from fatty tissues to other tissues during fish preparation and cooking processes (e.g., U.S. EPA, 2000). Thus, the analysis of raw fillets (muscle tissue) with the skin and fat removed may not have been representative of potential PCB contamination in local fish (if present). However, it is noted that PCB concentrations in surface water have not been detected to date, and as such, PCB concentrations in fish (associated with the PCB Area) would not be anticipated.

- In the HHRA, it is assumed that PCB bioavailability to human receptors is the same as in the toxicology studies that the route-specific TRVs were derived from. Exceptions to this were made for dermal exposure pathways and routes. This assumption likely overestimates exposure as it is well known that the bioavailability of most contaminants in a soil matrix is reduced relative to that in foods or drinking water (which are the exposure media typically used in toxicology studies).
- Considering PCB soil concentrations from depths down to 1.5 m represent surface soil is a conservative assumption that overestimates human contact with PCB-impacted soil. While 1.5 m is the standard surface/subsurface soil depth cutoff from CCME, in reality, the most relevant soil depth with respect to direct human contact with soil (assuming no digging or excavation occurs) is the top 5 to 15 cm.
- There is some uncertainty regarding the status of PCBs as potential human carcinogens. While PCBs were assessed as non-carcinogens in the HHRA (which is considered most appropriate at this time, given the current Canadian TRVs for PCBs and the exposure conditions associated with the Site), consideration was given to how risk estimates would change under the assumption that PCBs are carcinogens (and using cancer-based TRVs).
- The HHRA assumed the human health effects of PCBs via inhalation are the same or similar to those which occur via the oral route of exposure. Although there is limited information in the scientific literature to support this assumption, it is believed to be a conservative assumption that overestimates the potential for PCBs to result in human health risks.
- The PCB media and biota chemistry data collected to date are expressed as total PCBs (with Aroclor resemblance also reported for samples with detectable PCB concentrations). No data have been collected to date for specific PCB congeners, including the "dioxin-like" coplanar PCB congeners. Given that burning occurred in some areas of the WDS (although, apparently not within the PCB Area), and that dioxins and furans have been measured in environmental media of the WDS, consideration could be given to conducting PCB congener analysis in a future monitoring event to determine if there are elevated media and/or biota concentrations of coplanar PCB congeners that would be expected to contribute to potential effects related to the co-presence of dioxin and furan congeners. At this time, it is not possible to evaluate the potential contribution of coplanar PCBs to the effects that may be caused by the presence of dioxins and furans in WDS media.

9.0 ECOLOGICAL RISK ASSESSMENT

9.1 ERA Goals, Approach and Scope

In many ERA frameworks, it is common to establish management and assessment goals for the ERA of a given site. The recent Federal Contaminated Sites Action Plan (FCSAP) ERA guidance (Azimuth, 2012a) defines a site management goal as the overall planning objective for a site, which provides a statement about the desired condition of an ecosystem or its components within the context of current or future site use. The site management goal for this ERA was to determine whether or not environmental media at the Site (i.e., PCB Area of the WDS) warrants risk management for the protection of ecological receptors that may occur on or use the Site. The assessment goal for an ERA relates to the management goal. In this ERA, the assessment goal was to determine if there are potential unacceptable ecological risks associated with Site conditions, and if so, determine what (if any) management or remedial action is needed to mitigate the risk.

The scope of this ERA is limited to the on-site terrestrial media and surrounding off-site terrestrial and aquatic media. The scope of the ERA is also limited by the chemistry data currently available for the on- and off-site media. The ERA does not evaluate or consider potential exposures and risks to ecological receptors that may occur on adjacent properties other than the WDS, nor does it consider the freshwater environments of water bodies other than those that appear to have a potential to be influenced by the PCB Area in general.

9.2 ERA Framework

The ERA was conducted using widely accepted ERA frameworks, methodologies and guidance published and endorsed by Environment Canada (*e.g.*, CCME, 1996; the recent FCSAP ERA Guidance document (prepared for Environment Canada by Azimuth, 2012a)) and the United States EPA (1998). Where necessary, consideration was also given to ERA guidance from other agencies and jurisdictions in Canada and internationally. Where deemed necessary, such guidance is cited within the appropriate sections of this report.

ERAs can involve various levels of detail, complexity and level of effort. The ERA framework developed by Environment Canada (CCME, 1996) reflects this in its tiered approach, where each successive tier is sequentially more detailed than the previous one, with assessment characteristics ranging from a simple, qualitative and literature-based approach for the first tier (often termed Screening Level Assessment) to complex, predictive and field-based approaches for the second and third tiers (often termed Preliminary Quantitative and Detailed Quantitative Assessments, respectively). While moving from one tier to the next increases the complexity and effort of the ERA, this is typically only required if the results of the previous tier indicate a more complex and detailed assessment is warranted. The more recent ERA

guidance endorsed by Environment Canada for use on federally contaminated sites (i.e., Azimuth, 2012a) does not categorize ERAs according to scope or level of detail/effort (e.g., screening-level versus detailed quantitative). Rather, this recent ERA guidance suggests the level of detail and effort for an ERA is dependent on many factors and is often site and assessment-specific. This guidance also recognizes the level of detail and effort in an ERA may involve a combination of qualitative screening level and quantitative approaches, depending on the chemicals and receptors selected for assessment, as well as, spatial and temporal factors or boundaries associated with the ERA.

In general, if the use of conservative assumptions related to both chemical exposure and toxicity to ecological receptors (as would be commonplace in an initial tier of an ERA) indicates a low potential for ecological risks, there is typically a high degree of confidence in this finding, such that areas and/or receptors with a low risk potential may be excluded from further assessment. However, in situations where the initial tier of an ERA indicates an elevated potential for ecological risk or identifies key data gaps that preclude the completion of an ERA, further data collection is typically required and/or more detailed ERA approaches are typically applied (i.e., those that are less conservative but more realistic or site-specific).

The specific steps used to conduct this ERA are based on CCME (1996) and Azimuth (2012a) and are illustrated in Figure 9-1.

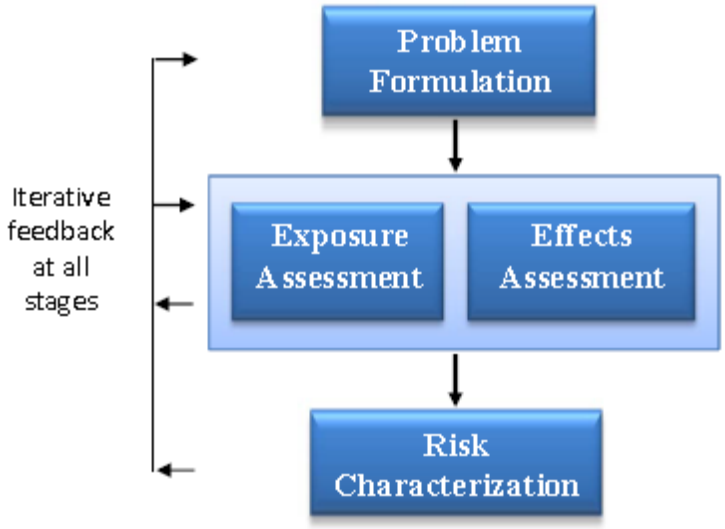


Figure 9-1 Ecological Risk Assessment Steps (Azimuth, 2012a)

Each of these steps of ERA are briefly described below, with further details provided in subsequent sections.

Step I: Problem Formulation: Similar to HHRA, the problem formulation step of an ERA acts as an information-gathering and interpretation stage, which serves to plan and focus the approach of the ERA on the most critical areas of concern for the site being evaluated. There are several elements or tasks that comprise the problem formulation in an ERA, including:

- site characterization and review of existing site information;
- review regulatory context;
- establish the objectives, goals and level of effort of the ERA;
- select study boundary and reference areas;
- identify receptors of concern;
- identify assessment and measurement endpoints;
- identify exposure pathways;
- identify COPCs;
- develop LOE; and
- develop a CSM for the ERA.

The outcomes of the problem formulation form the basis of the approach taken in the ERA. These outcomes can also help identify key data gaps that may limit aspects of the ERA, lead to developing sampling and analytical plans, and determine whether or not there is a need to further evaluate certain areas of a site, receptor types, exposure pathways and site-related chemicals.

Step II: Exposure Assessment: The exposure assessment step of an ERA involves estimating the amount of each chemical of concern that is potentially received by each selected ecological receptor or receptor group. For quantitative assessments, exposures are generally estimated using key receptor physiological and ecological characteristics and parameters (e.g., body weight, diet proportions, food intake rates, energy utilization, home ranges, amount of time spent in study area, etc.). For more qualitative (or screening level) assessments, receptor exposures are often assumed to be equal to measured media concentrations (e.g., soil concentrations are assumed to represent exposure concentrations for soil invertebrates).

Step III: Effects Assessment: Also called hazard or toxicity assessment in some jurisdictions, this step identifies TRVs or other types of toxicity benchmarks for each receptor or receptor group evaluated, for each COPC.

Step IV: Risk Characterization: In an ERA, risk characterization typically consists of the evaluating and interpreting each line of evidence (LOE) considered in the preceding steps of the ERA. This is often

done using a weight of evidence (WOE) approach to make conclusions on the probability and/or potential magnitude of ecological risk. Consideration of the various uncertainties, limitations and conservative assumptions within the ERA is also an important consideration in ecological risk characterization. The outcomes of risk characterization may lead to additional media or biota sampling, additional site characterization, risk management recommendations or corrective remedial action.

9.3 Problem Formulation Approaches and Outcomes

Characterization information for the Site, including the general surface coverings and habitat types present, has been previously discussed in Section 3.0.

With respect to the regulatory context for the ERA, the legislation and regulations referred to in Azimuth (2012a) are acknowledged and considered herein, as necessary. As noted in the HHRA (Section 8.0), the Site is considered to be classified as commercial land use.

The management and assessment goal for the ERA were previously presented in Section 9.1. The ERA of the Site is conducted primarily at a screening level and uses a mix of qualitative and quantitative approaches.

9.3.1 Selection of Site Boundaries and Reference Areas

The spatial boundary for this ERA is the boundary of the PCB Area of the WDS, as well as the areas bounded by sampling locations within off-site media that have been used to monitor the potential off-site migration of PCBs from the PCB Area. The property around the WDS is assumed to be owned by the Crown and is not within the scope of the ERA. Water bodies that are not potentially influenced by the WDS, or are not hydraulically connected to potentially influenced water bodies, are also not considered in this ERA.

The temporal boundary for the ERA is current and future conditions. No specific reference areas were selected for the ERA, other than sampling locations that were previously identified by others as being "background" areas or up-gradient of the PCB Area. For PCBs, the typical default condition for reference areas is the assumption that environmental media and biota concentrations are non-detectable.

9.3.2 Identification of Receptors of Concern

A receptor of concern (ROC) is any non-human individual, species, population, community, habitat or ecosystem that is potentially exposed to COPCs (Azimuth, 2012a). Consideration of potential ROCs for an ERA is site-specific and must reflect an understanding of the specific ecological attributes of the site being assessed. For example, sites that offer limited or no habitat or food resources for ecological receptors likely do not merit an ERA study. Identification of ROCs is not limited to those that only occur

on the Site, but also considers receptors that may use the Site for foraging, breeding, nesting, resting and other aspects of their life history.

For the purposes of ERA, it is neither practical nor necessary to assess each and every species that may potentially occupy or use a site. Instead, it is common practice to identify a selected subset of species as the ROCs for the assessment. There are many considerations when identifying ROCs for an ERA, which include the following:

- General site characteristics (e.g., surface coverings, habitat types present, observed species) as determined from ESAs;
- Representation from the various trophic levels, habitats and feeding guilds that are appropriate for a site;
- Behavioural and physiological characteristics that would increase or decrease the potential for chemical exposure (*e.g.*, diet and habitat preferences; feeding behaviour; home/foraging/breeding ranges; mobility; body weights, etc.); for example, species that are highly omnivorous are rarely selected as receptors in an ERA as it is much more difficult (due to high variability and uncertainty) to estimate the exposure such species may incur from food items, relative to species that are primarily in one feeding guild, such as herbivore, granivore, insectivore/invertevore, carnivore, etc.;
- Habitat quality, suitability and preferences for receptors that may occur on or use a site (e.g., does the site meet habitat requirements or preferences for receptors of interest?);
- Likely percentage of time spent within potentially impacted areas of a site and fraction of diet obtained from these areas;
- Whether species of interest are resident biota or migratory (and other seasonal factors, such as species that may hibernate);
- The availability of biological data describing receptor characteristics, life history and behaviour;
- The physical-chemical, environmental fate/behaviour and toxicological properties of COPCs (such as persistence and potential to bioaccumulate and/or biomagnify in terrestrial and/or aquatic food webs; known sensitivity of certain wildlife species to certain chemicals);
- Availability of reliable ecotoxicological data for the receptor or receptor group;
- Availability of appropriate measurement endpoints for the ROC;
- Availability of regional and local habitat surveys or species inventories;
- Potential or documented presence of species that are at risk (e.g., listed as rare or endangered) or have some similar status within provincial jurisdictions;
- Socioeconomic considerations (such as: is a species commercially important? Is a species valued by humans, or is it considered a pest or vermin?);
- Availability of information from local experts and residents of the area or surrounding properties;
- Potential presence of domestic animals (e.g., livestock, cats, dogs); domestic animals may have different protection goals or biological endpoints of interest than wildlife, and the presence of

domestic animals may serve as key ecological stressors for wildlife species that may occur at a given site;

- A site visit to visually confirm habitat types, exposure pathways and the potential for certain wildlife species to occur on or use the Site (a site visit was conducted by Dillon personnel in March 2013); and,
- Professional judgement.

The key outcome of the ROC Identification Step is the consideration of receptor types that could potentially be included in the ERA, along with rationale or justification for why certain receptor types are included or excluded from the ERA. This is often presented as a table or matrix format. Ideally, the ROCs selected for an ERA will be those that occur at the site (or would be expected to use the site on a regular basis), have a high exposure potential to COPCs in site media and/or have a known sensitivity to one or more COPCs. When the selected ROC meets these general conditions, the likelihood for the occurrence of adverse effects in less exposed or less sensitive receptors would be lower than for the assessed receptors.

To identify ROCs for the Site, a number of information sources were reviewed and considered including:

- Previous ESA and monitoring study reports (i.e., documents listed in Section 4.0).
- Azimuth (2012b), FINAL, Federal Contaminated Sites Action Plan (FCSAP) Ecological Risk Assessment Guidance, Module C: Standardization of Wildlife Receptor Characteristics. This document suggests a number of candidate receptors for Canadian federally owned sites.
- Information obtained from a site visit by Dillon personnel.
- Natureserve Explorer (http://www.natureserve.org/explorer/).
- U.S. EPA, 1993, Wildlife Exposure Factors Handbook, Volume I of II, Office of Health and Environmental Assessment, Office of Research and Development, U.S. Environmental Protection Agency, EPA/600/R-93/187.
- Arenal, C.A., Sample, B.E., Ziolowski, D.J., and Johnson, M.S, 2006, Development of Terrestrial Exposure and Bioaccumulation Information for Ecological Risk Assessments. Contract No. DAAD050-00-P-8365, U.S. Army Center for Health Promotion and Preventive Medicine, Toxicology Directorate, Health Effects Research Program.
- Specific guidance related to spatial and habitat quality considerations for ERA (i.e., ASTM, 2002; 2011).

Table 9-1 presents ecological receptors and surrogate species that were considered, and whether or not they were included or excluded from evaluation in the ERA along with rationale. Among the rationale for inclusion/exclusion of ecological receptors is the presence/absence of operable exposure pathways by which receptors could come into contact with PCBs in environmental media. While the selection of

exposure pathways is typically its own item within the Problem Formulation step of ERA, it is only conducted for ROCs that have been deemed to merit inclusion in the ERA. Furthermore, receptor and exposure pathway selection are linked processes that influence each other and it is commonplace for exposure pathway considerations to be a factor in deciding whether or not certain ecological receptors should be selected for evaluation in an ERA. Surrogate receptor species are also presented in Table 9-1, if/where relevant. As noted in Azimuth (2012a) and CCME (1996), surrogate receptor species are used to represent particular feeding guilds or ecological niches and are selected based on many of the same considerations used to identify ecological ROCs (as noted above).

Table 9-1 Ecological Receptors Selected for / Excluded from Evaluation in the ERA

Receptor Group	Include/ Exclude	Surrogate Species for Receptor Group (if included)	Rationale for Inclusion/Exclusion
Terrestrial vegetation	Exclude		While terrestrial vegetation would typically be considered in an ERA (as vegetation is in direct contact with soil and is immobile), this receptor group is excluded in this ERA. The reasons for excluding terrestrial vegetation reflect the use of the PCB Area and the rest of the WDS as an active waste disposal facility for decades. This permitted activity has clearly disturbed soils and resident native vegetation communities within the PCB Area over many years, such that any vegetation currently present within this area would be tolerant species that have colonized soils since the most recent disturbance. The process of PCB Area closure and installation of the engineered cap will eradicate any existing terrestrial vegetation communities currently occurring within the PCB Area. There is no merit in assessing or protecting existing ecological receptors when it is known that they will be removed or covered as part of site closure processes. Furthermore, the PCB Area will receive an engineered cap as part of the WDS closure process. This cap will comprise a depth of approximately 2.0 m of materials on top of existing PCB Area soils. Such a depth will prevent any new vegetation growth from contacting PCB-impacted soils within the PCB Area.
Soil invertebrates	Exclude		While soil invertebrates would typically be considered in an ERA (as these organisms are in direct contact with soils and are relatively immobile), this receptor group is excluded in this ERA. The reasons for excluding soil invertebrates are the same as for terrestrial vegetation and reflect the use of the PCB Area and the rest of the WDS as an active waste disposal facility for decades. This permitted activity has clearly disturbed soils and resident native soil invertebrate communities within the PCB Area over many years, such that any invertebrate communities currently present within this area would be tolerant species that have colonized soils since the most recent disturbance. The process of PCB Area closure and installation of the engineered cap will largely eradicate any existing soil invertebrate communities currently occurring within the PCB Area. There is no merit in assessing or protecting existing ecological receptors when it is known that they will be removed as part of site closure planning processes. Furthermore, the PCB Area will receive an engineered cap as part of the

Receptor Group	Include/ Exclude	Surrogate Species for Receptor Group (if included)	Rationale for Inclusion/Exclusion
			WDS closure process. This cap will comprise a depth of approximately 2.0 m of materials on top of existing PCB Area soils. Such a depth will prevent or greatly limit any new colonies or invertebrate assemblages from contacting PCB-impacted soils within the PCB Area.
Soil microorganisms	Exclude		While soil microorganisms play key roles in nutrient cycling within soils (as they are primary consumers of organic matter which convert soil nutrients into forms that are available for uptake by plants and higher trophic level organisms), their inclusion as ROCs in an ERA is problematic, even though soil quality benchmarks exist that use nutrient cycling parameters as the biological endpoints. There is high spatial and temporal variability in soils with respect to microbial community composition and tolerance to most contaminants. Reliable toxicity data is extremely limited and exposure-response relationships are unclear for most contaminants, especially those that can be used by certain microbial species as energy sources or terminal electron receptors in respiration. There are limited tools and approaches that reliably determine microbial exposure and responses to chemical stressors. The ecological relevance of microbial responses to contaminants is questionable, given the often high rates of functional redundancy in soil microbial communities. Basic understanding of microbial community structure and function in soils is limited.
Herbivorous small mammals	Exclude		While herbivorous small mammals may occasionally occur/forage on the PCB Area where they could be exposed to PCBs in soil via the consumption of vegetation and soil, the Site's past use as an active WDS, small size and limited habitat and food resources (relative to the areas surrounding the WDS) is considered to limit the presence of these receptors, such that there would not be significant PCB exposures incurred by populations of herbivorous small mammals in relation to the PCB Area. The exclusion of these receptors is supported by the current limited vegetation present on the PCB Area, and given that future vegetation growth on the PCB Area (post-capping) will not be in contact with the PCB-impacted soils (>2.0 m depth to soil PCB impacts). Furthermore, it is well known that PCBs do not readily transfer from soils to terrestrial plants via root uptake (which is the only plausible soil to plant pathway that is operable in PCB Area soil) (e.g., ATSDR, 2000; CCME, 2001; U.S. EPA, 1997; 1999; 2005). Because of this, in most ERAs of PCBs, it is recognized that PCB uptake into soil invertebrates and consumption of invertebrates by invertevorous organisms is a pathway of greater ecological concern than PCB uptake into vegetation, and subsequent consumption of vegetation by herbivores.
Insectivorous or invertebrate-consuming small mammals	Include	Masked Shrew (*Sorex cinereus*)	Insectivorous/invertevorous small mammals have a high potential for exposure to chemicals in soils due to their burrowing behaviour, consumption of soil dwelling organisms and small home ranges. While the Site's past use as an active WDS, small size, and limited habitat and food resources (relative to the areas surrounding the WDS) is considered to limit the presence of these receptors, such that there would not be significant PCB exposures incurred by populations in relation to the PCB Area, it is well documented that PCBs are bioaccumulative substances with a

Receptor Group	Include/ Exclude	Surrogate Species for Receptor Group (if included)	Rationale for Inclusion/Exclusion
			potential to biomagnify in terrestrial food webs. In addition, consumption of invertebrates by invertevorous or insectivorous mammals is a known pathway of ecological concern in relation to PCBs (U.S. EPA, 1997; 1999; 2005). Although population level risks would not be expected on the basis of spatial and habitat quality considerations, it is possible that PCBs could accumulate within insectivorous mammals and potentially their predators as well. Thus, insectivorous/invertevorous small mammals (represented by the masked shrew) were conservatively included in the ERA.
Carnivorous small mammals	Include	Ermine - also known as short-tailed weasel (M*ustela ermine*)	Carnivorous small mammals may be exposed to PCBs via the ingestion of prey items that occur or forage on the PCB Area, and incidental soil ingestion from the PCB Area. Shrews are a common prey item of the ermine. While the Site's past use as an active WDS, small size, and limited habitat and food resources (relative to the areas surrounding the WDS) is considered to limit the presence of these receptors, such that there would not be significant PCB exposures incurred by carnivorous small populations in relation to the PCB Area, it is well documented that PCBs are bioaccumulative substances with a potential to biomagnify in terrestrial food webs. In addition, consumption of invertebrates by invertevorous or insectivorous mammals, and subsequent predation of such invertevores/insectivores is a known pathway of ecological concern in relation to PCBs (U.S. EPA, 1997; 1999; 2005). Although population level risks would not be expected on the basis of spatial and habitat quality considerations, and typical home/foraging range sizes, it is possible that PCBs could accumulate within the prey items of carnivorous small mammals and potentially within small mammalian carnivores themselves. Thus, carnivorous small mammals (represented by the ermine) were conservatively included in the ERA.
Herbivorous large mammals	Exclude		The exposure potential for large herbivorous mammals to PCBs in PCB Area environmental media would be extremely limited, given their large home/foraging ranges, as well as the Site's past use as an active WDS, small size, and limited habitat and food resources (relative to the areas surrounding the WDS). Furthermore, as described above for herbivorous small mammals, the exclusion of large herbivores is supported by the current limited vegetation present on the PCB Area, and given that future vegetation growth on the PCB Area (post-capping) will not be in contact with the PCB-impacted soils (i.e., >2.0 m depth to soil PCB impacts). Also, PCBs do not readily transfer from soils to terrestrial plants via root uptake.
Carnivorous large mammals	Exclude		The exposure potential for large carnivorous mammals to PCBs in PCB Area environmental media and prey items that forage on the PCB Area would be extremely limited, given their large home/foraging ranges, as well as the Site's past use as an active WDS, small size, and limited habitat and food resources (relative to the areas surrounding the WDS). It is considered that inclusion of the ermine within the ERA, which has a smaller home/foraging range and smaller body weight relative to large carnivores that may occur within Newfoundland, captures any potential risks that the PCB Area may pose to large mammalian carnivores.
Insectivorous and invertebrate-consuming birds	Exclude		While insectivorous/invertevorous birds may be exposed to PCBs by the consumption of soil invertebrates and by incidental soil ingestion (particularly if the birds are ground foragers and have small home or foraging ranges), the Site's past use as an active WDS, small size, and limited habitat and food resources (relative to the areas surrounding the WDS) is considered

Receptor Group	Include/ Exclude	Surrogate Species for Receptor Group (if included)	Rationale for Inclusion/Exclusion
			to limit the presence of these receptors, such that there would not be significant PCB exposures incurred by populations in relation to the PCB Area. As birds in this feeding guild tend to have larger home/foraging ranges than the shrew, higher body weights than the shrew, and do not burrow (and thus, would be less exposed to PCB-impacted soils), it is considered that any potential risks that the PCB Area may pose to insectivorous/invertevorous birds would be captured by the inclusion of the shrew in this ERA.
Herbivorous birds	Exclude		While herbivorous birds may be present in the areas surrounding the PCB Area, they would not be expected to occur or forage on the PCB Area, given their relatively large home/foraging ranges, as well as the Site's past use as an active WDS, small size, and limited habitat and food resources (relative to the areas surrounding the WDS). Furthermore, as described above for other herbivorous receptors, the exclusion of herbivorous birds is supported by the current limited vegetation present on the PCB Area and given that future vegetation growth on the PCB Area (post-capping) will not be in contact with the PCB-impacted soils (i.e., >2.0 m depth to soil PCB impacts). Also, PCBs do not readily transfer from soils to terrestrial plants via root uptake.
Carnivorous birds	Exclude		The exposure potential for carnivorous birds to PCBs in PCB Area environmental media, and prey items that forage on the PCB Area, would be extremely limited, given their large home/foraging ranges, as well as the Site's past use as an active WDS, small size, and limited habitat and food resources (relative to the areas surrounding the WDS). It is considered that inclusion of the ermine within the ERA, which has a smaller home/foraging range, and smaller body weight relative to many raptors and other predatory birds that may occur within Newfoundland, captures any potential risks that the PCB Area may pose to carnivorous birds.
Waterfowl	Excluded		As there is no evidence to date that off-site aquatic areas potentially influenced by the PCB Area have measurable PCB concentrations in aquatic media or biota, there is no need to consider waterfowl species (which forage primarily within water bodies) in this ERA. Waterfowl would not be expected to occur on or use the PCB Area for foraging, breeding or resting purposes.
Piscivorous (fish-eating) wildlife (birds and mammals)	Excluded		As there is no evidence to date that off-site aquatic areas have measurable PCB concentrations in aquatic media or biota (i.e., fish), there is no need to consider fish-eating wildlife species in this ERA. Piscivores (mammalian or avian) would not be expected to occur on or use the PCB Area for foraging, breeding or resting purposes.
Amphibians	Excluded		While amphibian species may be among the most highly exposed and sensitive receptors to contaminants in environmental media, due to various aspects of their life history, dietary preferences and physiology, the availability of reliable or relevant toxicity data for these receptors is extremely limited to non-existent for most substances. Terrestrial exposure-based studies are particularly lacking. The general lack of suitable toxicity data precludes the assessment of amphibians in most ERA studies. Furthermore, the Site's past use as an active WDS, small size, and limited habitat and food resources (relative to the areas surrounding the WDS) is considered to limit the presence of amphibians, such that there would not be significant PCB exposures incurred by populations of these receptors. The PCB Area does not contain vernal pools or other habitat features (such as

Receptor Group	Include/ Exclude	Surrogate Species for Receptor Group (if included)	Rationale for Inclusion/Exclusion
			dead logs, rocks, etc.) that would encourage the presence of terrestrial amphibians (or terrestrial life stages of aquatic amphibians). Furthermore, with respect to aquatic amphibians (or aquatic life stages), there is no evidence to date that off-site aquatic areas have measurable PCB concentrations in aquatic media or biota. As such, there is no need to consider aquatic amphibians in this ERA.
Reptiles	Excluded		While reptile species may be among the more highly exposed and sensitive receptors to contaminants in environmental media, due to various aspects of their life history, dietary preferences and physiology, the availability of reliable or relevant toxicity data for these receptors is essentially non-existent at this time for most substances. This paucity of suitable toxicity data precludes the assessment of reptiles in most ERA studies. Furthermore, terrestrial reptile species do not occur in Newfoundland (See: http://www.env.gov.nl.ca/env/wildlife/all_species/vertebrates.html).
Rare, threatened, endangered, or "at risk", species	Excluded		None of the past ESAs or monitoring programs for the PCB Area and WDS addressed the potential presence of these types of species. Based on review of Species at Risk information provided by GNLDEC (i.e., http://www.env.gov.nl.ca/env/wildlife/endangeredspecies/index.html) and related links), the Atlantic Canada Conservation Data Centre, Natureserve Explorer (http://www.natureserve.org/explorer/), and the SARA Registry (http://www.sararegistry.gc.ca/default_e.cfm), there does not appear to be any "at risk", special concern, endangered, threatened or vulnerable species that are likely to occur on or in the vicinity of the PCB Area or the WDS, to any significant extent. For the majority of such species, they are either not known to occur in the vicinity of the WDS, or would be unlikely to occur or forage on or near the WDS due to their habitat and dietary preferences. While some of these species may occasionally occur near the WDS, they would not be expected to spend sufficient time at this location to incur significant exposures to PCBs from PCB Area soils.

The level of biological organization at which a ROC is evaluated in an ERA is an important concept that links closely with ecological protection goals for the ROCs. For lower trophic levels, such as vegetation, soil and sediment invertebrates and pelagic aquatic life (e.g., fish, invertebrates), the level of biological organization evaluated in an ERA is generally community level (Azimuth, 2012a; Suter et al., 2000). The community level is also considered the relevant level of biological organization when a receptor group has limited ecotoxicity data available (e.g., amphibians and reptiles). For higher trophic level receptors (such as birds and mammals), the ROCs are usually evaluated at the population level of biological organization. The individual organism level of biological organization is typically evaluated in an ERA only if the ROCs are rare, threatened or endangered species (Azimuth, 2012a; Suter et al., 2000). Thus, for most ROCs assessed at contaminated sites, the relevant level of biological organization is either community or population. The ecological protection goals for ROCs are the same in that the goal of an ERA is to protect most ROCs at the population or community level (unless there is evidence that the ROCs being evaluated are rare, threatened, endangered or listed as "species at risk" under the Species at Risk Act (SARA). As such, in most contaminated site ERAs, the focus is not on protecting individual organisms or even groups or individuals (such as breeding pairs) that may occur on or use a site. Rather,

the ecological protection goal is focused on maintaining local populations of the ROC (or its surrogate) or community ecological structure and function.

Freshwater aquatic receptors are not included in Table 9-1. This is because all of the freshwater aquatic receptors (or receptor groups) that were considered (such as pelagic and benthic invertebrates, fish) were excluded from further evaluation on the basis that to date, sampling and analysis of aquatic media and biota, at a number of locations down-gradient of the PCB Area, and in the direction of WDS surface runoff and groundwater flow, has indicated PCBs were not detectable (i.e., <RDL), with one exception (i.e., a single groundwater sample from MW-01, for the October, 2009 sampling event). More specific rationale for the exclusion of freshwater aquatic ecological receptors is provided in the following bullets. Summaries of the available aquatic media chemistry data are presented in Table 5, Appendix D.

- Eighteen surface water samples collected over eight sampling events from 2002 to 2011, at various locations (including "The Steady", Three Corner Pond, Denny's Pond, Gull Pond (which is up-gradient of the WDS), the leachate collection pond and the stream located immediately down-gradient of the WDS) have consistently had non-detectable total PCB concentrations (i.e., <RDLs). The RDLs have ranged from <0.04 to <0.06 µg/L across the surface water sampling and analytical programs that have been conducted.
- Water samples collected from test pits in 2002 (SGE, 2003a) indicated non-detectable concentrations of total PCBs (RDLs ranged from <0.05 to <0.5 µg/L).
- A ditch water sample (from the ditch adjacent to the PCB Area) collected in March 2009 (reported in SNC, 2012) indicated a total PCB concentration of 0.05 µg/L, which is equal to the typical RDL. It is likely that PCBs were not actually quantified in this sample considering typical laboratory measurement error and the possibility this particular water sample may have contained some suspended soil particles. Any water that collects in this ditch flows to the leachate collection pond.
- Six aquatic sediment samples collected over four sampling events from 2002 to 2010 at various locations (including "The Steady", Gull Pond (which is up-gradient of the WDS), the leachate collection pond and the stream located immediately down-gradient from the WDS) have consistently had non-detectable total PCB concentrations (i.e., <RDLs). The RDLs have ranged from <0.01 to <0.05 µg/g across the sediment sampling and analytical programs that have been conducted.
- Three ditch samples labelled as "sediments" by AMEC (2010) are more properly considered soil samples and were treated as such in the ERA (and the HHRA). PCB concentrations were <0.05 µg/g in two of the three samples, and 2.8 µg/g in one sample.
- Fifty-two groundwater samples collected over nine sampling events from 2007 to 2011, at eight monitoring well locations surrounding the PCB Area (including locations down-gradient of the PCB Area in the direction of groundwater flow) have consistently had non-detectable total PCB concentrations (i.e., <RDLs), with one exception. The RDLs have ranged from <0.04 to <0.06 µg/L across the groundwater sampling and analytical programs that have been conducted, for the most part. One groundwater sample from MW-03 in February 2007 had an elevated RDL for total PCBs due to

non-PCB peaks masking the chromatogram. The PCB concentration in this sample was <0.4 µg/L. The single exception to non-detectable PCB concentrations in groundwater samples was for MW-01 in October 2009 where the total PCB concentration was reported as 0.07 µg/L. All subsequent samples collected from MW-01 (three additional events) had non-detectable PCB concentrations (i.e., <0.05 or <0.06 µg/L). It is considered likely that the measured value of 0.07 µg/L is an analytical artefact as it is very close to the typical RDL value for PCBs in groundwater, and is well within typical laboratory measurement error. Furthermore, 0.07 µg/L is below the OMOE (2011) GW-3 value for total PCBs with shallow soil, coarse soil texture and non-potable groundwater conditions (i.e., 0.14 µg/L). All of these conditions are representative of those that occur at the WDS. This GW-3 value is the only applicable groundwater benchmark identified for total PCBs that is for the protection of freshwater aquatic life.

- Fish samples have been analyzed for total PCBs in two sampling events. The first event analyzed dead sticklebacks from Denny's Pond (SGE, 2003a,b). However, details were not reported regarding whether or not specific tissues were analyzed or whole fish, or even the number of fish samples that were analyzed. All that was reported is that dead stickleback sample(s) contained non-detectable concentrations of total PCBs. AMEC (2007a) reported that ten Brook Trout fillet samples, of a size that is typical for human consumption (i.e., at least 160 mm in length), analyzed raw with skin and bones removed, contained non-detectable concentrations of total PCBs (RDL was <0.005 µg/g dry weight). The fish were collected in September and December of 2006 from the streams located south of the WDS (prior to inflow into Loo and Three Corner Ponds). Ten fish were also collected from the Gull Pond outflow stream to represent a local reference stream. Gull Pond is up-gradient of the WDS. All reference Brook Trout fillet samples (same size range, and same sample preparation and analyses as the "site" trout samples) also contained non-detectable concentrations of PCBs.

Review of the laboratory data QA/QC information for the aquatic media PCB results indicated most data were of acceptable quality for risk assessment purposes. However, groundwater and surface water data collected during the December 2011 sampling event (SNC, 2012) indicated poor performance with respect to standard laboratory QA/QC acceptance limits for a few QA/QC performance measures. These data indicated elevated RDLs and surrogate recoveries outside acceptance limits. In addition, some of the submitted water samples contained sediment, some samples lacked sufficient sample volume and some samples were submitted past the recommended hold time for total PCB analysis. While the reported PCB concentrations in the affected samples indicated the same trend as all previous monitoring events (i.e., PCB concentrations <RDLs), the data must be viewed with some skepticism given these data quality issues.

Further perspective related to the inclusion and exclusion of mammalian, avian and amphibian receptors for the ERA of the PCB Area is provided below.

Size of PCB Area and Available Portions of the Site that Comprise Potential Habitat

It has long been recognized in ERA guidance and literature that consideration of the spatial scale of impacts at a given contaminated site can be useful in determining if potentially significant exposure conditions exist for ecological receptors, and for determining if risk estimates are ecologically significant (U.S. EPA, 1997; 1998; 1994; American Society for Testing and Materials (ASTM) 2002; MCP, 1996). Ideally, spatial scale issues are discussed at the onset of the ERA process. This can allow small sized sites that do not/may not require ERA to be excluded early in the ERA process, such that resources are not needlessly allocated to the assessment of these sites. ASTM (2002) notes the space or size of a contaminated site or area is directly related to the potential for ecological receptor exposure. Consideration of spatial scale can also help focus an ERA on the issues or receptors of greatest ecological relevance and/or provide a basis for determining an ERA for a given site may not be necessary, for some or all receptors of interest.

ASTM (2002) suggests for terrestrial environments (or sites), ecological habitat areas of less than two acres (equivalent to approximately 0.81 ha) are commonly considered small enough to not require ERA, so long as there are no site-specific issues of special concern. While it is noted that sites of <2 acres may contain foraging or breeding areas for individual small mammals, birds, and herpetofauna (amphibians and reptiles), this spatial scale does not usually support local populations of such wildlife species, and thus, would not requiring an ERA in most cases. Vegetation and soil invertebrates are typically excluded from this criterion, as there can be populations or communities of these receptors on sites <2 acres in size so long as the habitat and soil conditions are suitable. ASTM (2002) also notes that site coverings and surface features (such as foundations, pavement, gravel, concrete, fences, walls) are important to account for, as they limit the size of the habitat and foraging areas for various receptors.

The PCB Area has a total area of approximately 913 m^2 (or 0.091 ha) (AMEC, 2013). The total areal extent of the PCB Area is less than the ASTM (2002) spatial criterion of 0.81 ha, which suggests the PCB Area would not support local populations of mammalian, avian or amphibian receptors. Moreover, not all portions of the PCB Area have soil PCB impacts, thus the area available for potential PCB exposure to ecological receptors is actually less than 0.091 ha.

Consideration of Habitat Quality and Habitat Preferences

ASTM (2011) provides recent guidance on using habitat quality and preference information as a means of determining whether or not an ERA of a site is warranted. This guidance defines "habitat" as the combination of physical (landscape) and biological features preferred by a particular species. ASTM (2011) notes that consideration of landscape features to characterize habitat quality can enhance the ecological relevance of an ERA, and avoid conducting ERAs on sites or in areas where wildlife species would be absent (or limited) because of a lack of suitable habitat. ASTM (2011) further notes that even if

the habitat is determined to be suitable for certain ecological receptors, other factors such as limited food resources, predation, or human disturbances may override the apparent habitat quality for a given receptor, such that its exposure to contaminants in site media and biota would be minimal. Because the foraging behaviour/time and occurrence of receptors on a site is directly related to habitat suitability, the exposure potential to contaminants in site media is also directly related to the suitability of the site's habitat for a given receptor. ASTM (2011) notes that ERAs should only be conducted for sites or areas where ecological receptors occur, or would likely occur. If receptors are not present or are unlikely to be present because of poor habitat quality or suitability, then an ERA of those receptors is not warranted.

Other habitat-related considerations suggested by ASTM (2011) include the following:

- Consider the size of the site relative to receptor home/foraging/breeding range requirements.
- Consider the site habitat quality relative to habitat quality on adjacent or surrounding sites. If a site's habitat quality is approximately equal to that of the site surroundings, the proportion of time that an animal will spend on the site will likely be proportional to the surrounding sites and bounded by the size of the animal's home range. If the habitat on the site is of lower or higher quality than the surrounding sites, then an animal is likely to spend proportionally less or more of its time on the site.

To further explore these concepts in relation to the Site, information was sourced on habitat preferences and home/foraging/breeding range size for the selected ecological receptors in the ERA (i.e., masked shrew, ermine) and also for other representative small mammal and avian receptors that are commonly evaluated as ROCs in ERAs within Atlantic Canada (i.e., Snowshoe Hare, Red Fox, American Robin). The latter three receptors are used for illustrative purposes only in Table 9-2.

Review of habitat preferences for these receptors (from information provided in: Azimuth, 2012a,b; U.S. EPA, 1993; Arenal et al., 2006; and, Natureserve Explorer (http://www.natureserve.org/explorer), indicates Site conditions are not consistent with their preferred habitat types. Thus, it can be concluded the Site does not offer significant suitable habitat conditions for common small mammal and avian receptors. While this does not mean that individual animals (out of these receptor groups) will not periodically occur on, or use the Site, the probability that local populations of these receptors would use the Site (for foraging, nesting, breeding etc.) is considered to be very low. This conclusion is supported by conducting some example calculations for N_s for these receptors (where N_s is the number of individuals of a given receptor species that are likely to inhabit any habitat subdivision on a site), as presented in ASTM (2011). The N_s calculation is similar to the common Area Use Factor (AUF) calculation that is sometimes used in ERAs. One form of this calculation is as follows (ASTM, 2011 also provides some alternate means of determining N_s).

$$N_S = \frac{A_S}{HR_S}$$

Where,

N_s = the number of individuals likely to inhabit the habitat subdivision on a site and incur potential exposures to COPCs from site media. In this case, the habitat subdivision of interest is the total spatial area of the PCB Area.

A_s = the area of the habitat subdivision (or portions of the site with potential habitat or foraging areas); 0.091ha for the Site.

HR_s = the approximate home/foraging range size of the receptor (ha); based on literature regarding lowest reported home range sizes.

Example N_s calculations for the Masked Shrew, Ermine (which are evaluated in the ERA), and Snowshoe Hare, Red Fox and American Robin (not evaluated in the ERA) are presented below (Table 9-2). For all receptor species, the HR_s value was taken from recent FCSAP ERA guidance documentation (i.e., Azimuth, 2012b). The A_s value is 0.091 ha, as noted above.

Table 9-2 Estimated N_s Values for Selected Ecological Receptors

Receptor	A_s	HR_s	N_s
Masked Shrew	0.091ha	0.6 ha	0.15
Ermine	0.091 ha	1.0 ha	0.09
Snowshoe Hare	0.091 ha	1.6 ha	0.06
Red Fox	0.091 ha	280 ha	0.0003
American Robin	0.091 ha	0.7 ha	0.13

Given the estimated N_s values, it is likely that only a few (if any) individuals representing small mammal and avian receptors would be expected to occur on, or use the Site to any significant extent. While the N_s calculation is not without its uncertainty, it provides a reasonable indication of the numbers of individual organisms that may be expected to use a habitat subdivision on a site. Such small numbers for N_s are clearly not indicative of populations of these receptors. These calculations further demonstrate the Site would offer limited suitable habitat for supporting small mammals and birds and that the use of the Site by these receptors is likely to be limited.

When the Site habitat quality is considered relative to habitat quality on adjacent or surrounding sites/areas, it is evident that the areas surrounding the Site offer better habitat suitability for small mammals and birds than the Site does. There is nothing to distinguish the Site from these other areas as offering unique or preferred habitat for any ecological receptor population or community. This further supports that use of the Site by mammalian and avian receptors would likely be limited.

Other Considerations

As noted previously, the PCB Area will be capped as part of WDS closure. While the Site conditions post-capping may be more amenable to ecological receptors as potential habitat, or a potential food resource/foraging area, the PCB-impacted soils within the PCB Area will not be accessible, as these soils will be buried under a minimum of 2.0 m of engineered cap materials.

Summary

Based on the ROC selection procedure described above, only the masked shrew and the ermine are identified as ROCs that warrant further evaluation in the ERA. A number of other ecological receptors were considered but excluded from the ERA for the reasons provided in the preceding sections and Table 9-1. Subsequent sections of the ERA focus on these two small mammal receptor species. The masked shrew and ermine are considered "worst case" ecological receptors that represent other small mammals and birds as well, as they would be expected to incur higher exposures to PCBs from Site soil than most other small mammals and birds would, on the basis of small body weights, small home ranges, their feeding preferences and burrowing behaviour.

It must be recognized though that the evaluation of the shrew and ermine in the ERA is highly conservative given the information presented above on spatial size and habitat quality considerations for the Site, which strongly suggests that neither individuals nor populations of these receptors are likely to use or access the Site to any significant extent. Thus, ecological risk values generated for these species are considered to be substantial overestimates of what is realistic. The shrew and ermine are evaluated in the ERA primarily to illustrate potential exposures and effects (or risks) in representative species with a high exposure potential, in recognition of the fact that PCBs have a high potential to bioaccumulate and biomagnify in terrestrial food webs. It is also important to recognize the estimated exposures and risks to the shrew and ermine are only possible under current Site conditions, and that potential PCB exposures for these species (and similar species that the shrew and ermine represent) will be negligible once the engineered cap has been installed over the PCB Area.

9.3.3 Selection of Assessment and Measurement Endpoints and Lines of Evidence

An assessment endpoint is defined as an explicit expression of what is to be protected, defined by an ecological entity (i.e., receptor or receptor group) and by a characteristic (Suter, 1989; U.S. EPA, 1998; Azimuth, 2012a). The characteristic is a specific attribute or property for the receptor that is important to protect and which is potentially at risk (*e.g.*, abundance, survival). As noted previously, the ecological entity (or receptor) can be defined at different levels of biological organization. An assessment endpoint must include a receptor (or receptor group) and a specific property or attribute of that receptor (Azimuth, 2012a). Assessment endpoints are quite similar to protection goals with the only notable difference being that the former describes the environmental attribute of interest, whereas the latter articulates the desired

state of that attribute (Azimuth, 2012a). It is common practice in ERA that assessment endpoints do not express a direction or desired state (such as: increased, decreased, healthy, sustainable, etc.)

Assessment endpoints may or may not be directly measurable (U.S. EPA, 1998). For example, the abundance of song birds may be assessed directly if avian surveys have been conducted, but would have to be assessed indirectly if survey outcomes are not available. If assessment endpoints are not directly measurable (which is not uncommon due to practical reasons), then other measures, called "measurement endpoints", may be used to evaluate the risk related to the assessment endpoints.

A measurement endpoint is considered to be any measure of exposure or effects for a ROC or any measure of change in the attribute of an assessment endpoint (Azimuth, 2012a). Measurement endpoints form the basis for LOE that are used to estimate risks in an ERA. Measurement endpoints and LOEs are developed at the same time. Similar definitions of measurement endpoints have been provided by others. For example, Suter II (1990) defined mmeasurement endpoints as responses to a chemical stressor that can be measured and quantified. CCME (1996) defines measurement endpoints as "the effects on an ecological component that can be measured and described in some quantitative fashion".

The U.S. EPA (1998) identifies three categories of measurement endpoints:

1. Measure of Exposure: a measure of chemical presence and movement in the environment and its contact with the receptor (e.g., concentrations of chemicals in soil).

2. Measure of Effect: a measure that describes a change in a characteristic of a receptor in response to a chemical to which it is exposed (e.g., laboratory aquatic toxicity test data).

3. Measure of Ecosystem and Receptor Characteristics: measures that influence the behaviour and location of receptors, the distribution of a chemical, and life-history characteristics of the receptor that may affect exposure or response to the chemical (e.g., home range and habitat requirements and preferences for a receptor).

A key consideration in the selection of measurement endpoints is how well a measurement endpoint represents an assessment endpoint and its ecological relevance. The greater the strength of association between the measurement and assessment endpoint, the greater the weight that is given to that measurement endpoint in the overall ERA, so long as the measurement endpoint is considered ecologically relevant.

Azimuth (2012a) defines LOE as any pairing of exposure and effects measures (or measurement endpoints) that provide evidence for the evaluation of a specific assessment endpoint. It is not uncommon for a LOE to use more than one measurement endpoint.

Essentially, measurement endpoints are tools, and LOEs are the way these tools are used and applied in the ERA. LOEs are directly related to both measurement endpoints and assessment endpoints. Four main categories of LOE are described in Azimuth (2012a), as follows:

- Site-specific toxicological evidence – Considers measurement endpoints related to studies of test organism exposures to contaminated site media under controlled conditions.
- Indirect toxicology evidence – Considers toxicological information obtained from other sites or literature, assuming the concentration-response relationships between sites is similar.
- Site-specific biological evidence – Considers direct assessment of the site biological conditions.
- Indirect biological evidence – Considers indirect assessment of biology, through extrapolation of knowledge obtained at other sites and from literature.

The Azimuth (2012a) guidance also identifies several criteria that are relevant to consider when selecting LOEs. For example:

- Ecological relevance – degree to which the assessment endpoint is represented by the LOE.
- Sensitivity – degree by which the LOE can detect change or differences from reference conditions.
- Specificity – degree to which the LOE is capable of distinguishing effects of site related COPCs from other factors and stressors.
- Spatial representativeness and site specificity – degree to which the LOE provides information that is site-specific and at a spatial scale relevant to the selected assessment endpoints.
- Temporal representativeness – degree to which the LOE captures temporal variation relevant to potential ecological risks.
- Expected data quality – degree to which the quality of data generated by the LOE will be acceptable (or not), such that the LOEs utility may be diminished.
- Expected acceptability – consideration of whether or not the LOE has standard test methods available or a long history of use that provides confidence and regulatory acceptance.

Specific assessment, measurement endpoints and LOE were identified for the ecological receptors evaluated in the ERA (i.e., masked shrew and ermine), which are presented in Table 9-3.

Table 9-3 Assessment and Measurement Endpoints and Lines of Evidence for Selected ROCs

Receptor Species	Assessment Endpoint	Measurement Endpoints	Lines of Evidence
Masked Shrew (*Sorex cinereus*)	Survival, growth, reproduction and abundance of populations.	Modelled exposure and risk estimates from PCBs in food items (assumed to be 100% soil invertebrates in this case) and soil. PCB mammalian TRVs. Spatial size and habitat quality/ preference considerations for the Site. Ns values estimated for the Site.	Comparisons between estimated PCB exposures and the TRV (i.e., ecological hazard quotients). Comparison of Site spatial size and habitat quality to ASTM (2002) spatial criteria for ERA, comparison of Site habitat quality to surrounding areas, comparison of receptor habitat preferences to Site habitat conditions and habitat quality. Interpretation of Ns values with respect to likelihood for populations or individuals to use or occur on the site.
Ermine - also known as short-tailed weasel (M*ustela ermine*)	Survival, growth, reproduction and abundance of populations.	Modelled exposure and risk estimates from PCBs in food items (assumed to be 100% shrews in this case) and soil. PCB mammalian TRVs. Spatial size and habitat quality/ preference considerations for the Site. Ns values estimated for the Site.	Comparisons between estimated PCB exposures and the TRV (i.e., ecological hazard quotients). Comparison of the Site spatial size and habitat quality to ASTM (2002) spatial criteria for ERA, comparison of Site habitat quality to surrounding areas, comparison of receptor habitat preferences to Site habitat conditions and habitat quality. Interpretation of Ns values with respect to likelihood for populations or individuals to use or occur on the Site.

Notes:
In the ERA context, the definition of a "population" can vary, and as such, very few ERA guidance documents define this term. In general, a population is a group of individuals of the same species that live together and breed amongst each other. Setting numerical limits on the number of individuals that comprise a population is inherently difficult and would vary greatly depending on the receptor species and its life history characteristics.
The abundance of a species could be affected directly (i.e., as a result of direct toxicity of COPCs on survival, growth or reproduction) or indirectly (e.g., as a result of decreased habitat suitability or reduced prey/food abundance) as a result of toxicity to food or prey items and other site activities that may affect habitat and presence of food resources. This ERA focuses on potential health effects as a result of direct chemical exposures.

9.3.4 Selection of Exposure Pathways and Routes

If there are no possible exposure pathways that link ROCs to chemicals of concern that are present in site media, there can be no potential for adverse effects from those chemicals. Therefore, it is important for any ERA to identify the major exposure pathways and routes for each of the selected ROCs (*i.e.,* masked shrew and ermine in the ERA). The difference between an exposure pathway and an exposure route has been previously described in Section 8.2.1 of the HHRA. The definitions and concepts noted in Section 8.2.1 apply equally to ecological receptors as they do to human receptors.

For mammalian receptors, it is common ERA practice to evaluate only those pathways that relate to the oral route of exposure (such as consumption of food/prey items, drinking water ingestion, incidental soil and sediment ingestion). Not only is the oral route the most commonly assessed exposure route in ERAs by far, but it is also almost always the dominant route that drives site-specific exposures and risks to ecological receptors at most sites. Dermal and inhalation-based exposure pathways rarely require evaluation in ERAs (Azimuth, 2012a, BC SAB, 2008; U.S. EPA, 2003). This is considered to be the case in this ERA as well, given the shrew and ermine both have fur that would limit dermal contact of PCBs with skin, and given PCBs are present within a soil matrix and any potential volatilization of PCBs from soil, or resuspension of Site soils in air would unlikely represent a significant source of PCB exposure relative to oral exposure pathways.

The main exposure pathways and routes evaluated for the masked shrew in the ERA are: soil ingestion and ingestion of soil invertebrates.

The main exposure pathways and routes evaluated for the ermine in the ERA are: soil ingestion, and ingestion of shrews. While ermine may occasionally consume soil invertebrates, such food items are not preferred relative to small mammals or birds (Azimuth, 2012b). Thus, it is assumed if ermine were forced to consume soil invertebrates as the primary food item at a given site, they would very likely relocate to areas where their preferred prey items are more abundant.

For both the shrew and the ermine, ingestion of drinking water is not considered in the ERA. This is because all aquatic sampling results to date have indicated no detectable PCB concentrations in surface water bodies that are potentially influenced by the PCB Area. Also, PCBs have been non-detectable in all but one groundwater sample (See discussion before Table 9-1 for additional details on PCBs in aquatic media). Furthermore, it is common for many small mammals to not consume water from surface water bodies, but rather, to obtain the majority of their water requirements from puddles or droplets on vegetation surfaces. The ingestion of aquatic sediments by the shrew and ermine is not considered in the ERA as these two receptor species forage almost exclusively in terrestrial areas, not aquatic areas. As noted, dermal and inhalation exposure pathways and routes are not considered for the shrew and ermine in the ERA.

With respect to the assessment and measurement endpoints and LOEs noted in Table 9-3, and the exposure pathways and routes selected for the shrew and ermine, there are some inherent conservative assumptions within the ERA that must be acknowledged. It is assumed that the shrew forages entirely in areas of the Site that contain PCB-impacted soils and its diet consists entirely of soil invertebrates. It is assumed the ermine diet is comprised entirely of shrews that forage in areas of the Site that contain PCB-impacted soils and that all soil exposure incurred by the ermine happens in areas of the Site that contain PCB-impacted soils. While these are typical assumptions for an ERA of a contaminated site, they are considered highly unrealistic and conservative.

9.3.5 Identification of COPCs for the ERA

As previously stated in Section 2.0, the current HHERA (of which the ERA is a component), is limited to the evaluation of PCBs only. Thus, PCBs are the sole COPCs for the ERA.

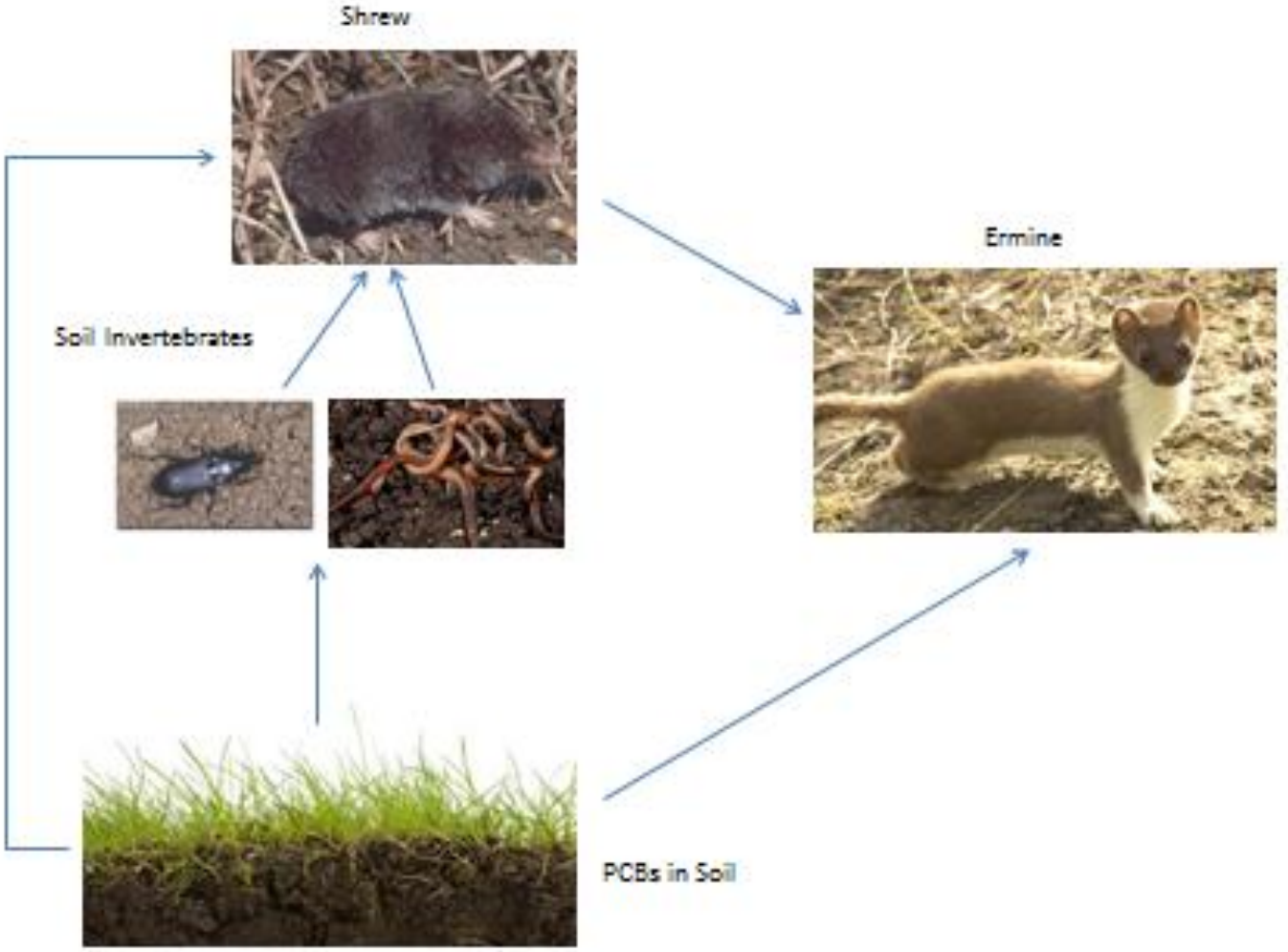
The discussion presented in Section 8.2.2 of the HHRA is considered to apply to the ERA as well.

9.3.6 Conceptual Site Model

A conceptual model is a written description and/or a visual representation of the relationships between the source of COPCs, the receiving environment(s) and the processes by which receptors may become directly or indirectly exposed to the COPCs (Barnthouse and Brown, 1994). Conceptual models can serve three purposes: 1) clarification of assumptions concerning the site or situation being assessed; 2) as a communication tool for conveying those assumptions; and 3) providing a basis for organization and completion of the ERA (Suter, 1999).

The conceptual model for the ERA is presented in Figure 9-2.

Figure 9-2 Conceptual Site Model for ERA



9.4 Exposure Assessment

The exposure assessment step of an ERA involves estimating the amount of chemicals that are received by ecological receptors.

Exposure can be calculated using quantitative approaches (e.g., where exposures of a specific receptor are estimated using models and a variety of receptor input parameters) or can be more qualitative in nature (e.g., where exposures are assumed to equal measured concentrations in environmental media). Estimating exposures using these latter methods likely overestimates potential exposure as it ignores an organisms' natural barriers to chemical uptake (i.e., bioavailability considerations) and biochemical transformation processes that may occur within cells, tissues and organs, which may reduce the actual dose that reaches a target site within the organism.

The degree of exposure of ecological receptors to chemicals in environmental media depends on the interactions of a number of parameters, including:

- The concentrations of chemicals in various environmental media (e.g., water, soil and food) as determined by the quantities of chemicals entering the environment from various sources, their persistence in these media and the normal ambient or background concentrations that exist independent of a specific source.
- The various exposure pathways for the transfer of the chemicals from the different environmental media to ecological receptors (e.g., inhalation of soil particles and dusts; ingestion of food items, water and soil/dust).
- The physiological and behavioural characteristics of ecological receptors that determine the actual exposures through interactions with the various pathways (*e.g.*, respiration rate, water intake, food intake and soil/dust intake).
- The various physical, chemical and biological factors that determine the ability of the ecological receptors to take the chemicals into their bodies from the exposure pathways (*e.g.*, bioavailability of the chemicals from soil/dust particles, foods, water and air).

For the masked shrew and ermine, potential PCB exposures were assessed quantitatively using site-specific assumptions, as well as standard food chain modeling equations, uptake factors (or equations), receptor physiological and behavioural parameters from regulatory agency ERA guidance documentation and/or scientific literature. The equations, factors, parameters and assumptions used in the exposure modelling are presented below in Section 9.4.2, along with the modelled PCB exposure estimates for the shrew and ermine.

The EPC used in the ecological exposure modelling was 27.1 mg PCBs/kg soil. This is the EPC value calculated from surface soil (i.e., ≤1.5 m) residual soil PCB concentrations, which is previously presented in Table 8.3. The EPC is the 95% Chebyshev (Mean, Sd) UCLM95 for these data (N=64).

9.4.1 ERA Exposure Scenarios

There are two exposure scenarios considered for the ERA: **1) Current Scenario (As Is – Outdoor Site Visitor (Trespasser))** and **2) Future Scenario (Post-Capping)**.

These scenarios were developed based on information previously presented in relation to site characterization, COPC selection, exposure pathway identification, and also considered the exposure scenarios developed for the HHRA.

As noted previously for the HHRA exposure scenarios, the WDS has a current commercial land use classification that is expected to continue for the foreseeable future, when the WDS is closed. There will not be any buildings or infrastructure or other site developments within the PCB Area other than the engineered cap as part of the closure process. There are no major barriers present now, or anticipated for the future, which would prevent or greatly restrict wildlife access to the PCB Area of the WDS.

The **Current Scenario (As Is – Outdoor Site Visitor (Trespassers))** ERA exposure scenario was evaluated quantitatively and assumes the shrew and ermine incur potential exposures to residual PCBs in PCB Area soil *via* the selected exposure pathways and routes (See Section 9.3.4).

The **Future Scenario (Post-Capping)** reflects the closure plan for the WDS, which involves placing an engineered cap over the PCB Area. This scenario determines if the cap is likely to minimize potential ecological exposures and risks, and was evaluated qualitatively based on such considerations as depth of the cap layers, the types of cap layers, the outcomes of the **Current Scenario (As Is – Outdoor Site Visitor (Trespasser))** and review of the environmental media and biota chemistry data that has been collected to date.

The assessment of ecological exposures and risks that may occur as part of future Site development scenarios is not within the scope of this ERA as there are no known plans for re-development of the Site or any portion of the WDS at this time.

Spatial boundaries for both ERA exposure scenarios are the current areal extent of the PCB Area (i.e., approximately 0.0913 ha). No other properties or off-site areas are considered in the ERA. Temporal boundaries for the ERA exposure scenarios reflect the scenarios themselves.

9.4.2 Equations, Factors, Parameters and Assumptions used in the Exposure Modelling for Masked Shrew and Ermine

Table 9-4 presents the receptor parameters and assumptions used to estimate PCB exposures to the masked shrew and ermine.

Table 9-4 Receptor Parameters and Assumptions for Masked Shrew and Ermine

Parameter/Assumption	Masked Shrew (*Sorex cinereus*)	Ermine (*Mustela erminea*)
Body Weight (kg)	0.0041 (average; Nagorsen, 1996)	0.089 (average; Raymond and Bergeron, 1986)
Expected Life Span (years)	<1; few to several months (Natureserve Explorer; URL: (http://www.natureserve.org/explorer/)	1-3 years on average (numerous sources)
Food Consumption Rate (g dry wt/day); Estimated using allometric equations from U.S. EPA (1993)	0.621 x (body weight (g))$^{0.564}$ = 1.38	0.235 x (body weight (g))$^{0.822}$ = 9.41
Home Range (ha)	0.6 ha (average; Nagorsen, 1996)	1.0 (Azimuth, 2012b)
Soil Ingestion Rate (% of food consumption rate)	13% of food consumption rate = 0.18 g dry wt/day (Sample and Suter, 1994)	2.8% of food consumption rate = 0.26 g dry wt/day (Sample *et al.*, 1997)
Percentage of Diet Obtained from Site (%)	100% (assumed)	100% (assumed)
Percent of Time Spent on Site (%)	100% (assumed)	100% (assumed)
Dietary Composition	mainly insects, earthworms, slugs, snails (U.S. EPA, 1993; Azimuth, 2012b)	mainly various small mammals, birds, amphibians, reptiles (U.S. EPA, 1993; Azimuth, 2012b)
Diet Assumptions for Exposure Modelling	100% soil invertebrates from the Site	100% shrew from Site
Oral Bioavailability of PCBs	1.0 (assumed)	1.0 (assumed)

The following equations illustrate how PCB exposures were estimated for the masked shrew and the ermine in the ERA, via the selected exposure pathways and routes.

PCB Uptake from Soil by Soil Invertebrates

$$C_{SInv} = C_S \times TF_{S-SInv}$$

Where:

C_{SInv} = calculated concentration in soil invertebrates (*183 µg/g dry wt. tissue*)

C_S = concentration in soil (*27.1 µg/g*; *UCLM95*)

TF_{S-SInv} = soil-to-soil invertebrate PCB biotransfer factor (*6.77 µg/g dry wt .tissue / µg/g soil*; U.S. EPA, 1999 – based on Aroclor 1254)

Thus, the calculated EPC of PCBs in soil invertebrates is 183 µg/g dry wt. tissue.

Masked Shrew PCB Exposures

Exposure via Soil Invertebrate Consumption

$$EXP_{FI} = \frac{C_{SInv} \times FI_{Shrew} \times D_O}{BW_{Shrew}}$$

Where:

EXP_{FI}	=	exposure from soil invertebrate ingestion (*6.2 x 10^4 µg/kg bw/day*)
C_{SInv}	=	concentration in soil invertebrates (*183 µg/g dry wt. tissue, as calculated above*)
FI_{Shrew}	=	food intake rate (*1.38 g/day dry wt.*)
D_O	=	fraction of diet from Site (*Assumed 1.0; unitless*)
BW_{Shrew}	=	body weight (*0.0041 kg*)

Thus, the calculated PCB exposure rate from consumption of soil invertebrates is 6.2 x 10^4 µg/kg bw/day.

Exposure via Soil Ingestion

$$EXP_{SI} = \frac{C_S \times SI_{Shrew} \times T_O}{BW_{Shrew}}$$

Where:

EXP_{SI}	=	exposure from ingestion of soil (*1183 µg/kg bw/day*)
C_s	=	concentration in soil (*27.1 µg/g*; *UCLM95*)
SI_{Shrew}	=	soil ingestion rate (0.*18 g/day dry wt.*)
T_O	=	fraction of time spent on Site (*Assumed 1.0; unitless*)
BW_{Shrew}	=	body weight (*0.0041 kg*)

Thus, the calculated PCB exposure rate from ingestion of soil is 1,183 µg/kg bw/day.

Total Exposure (from both soil and soil invertebrate ingestion)

$$EXP_{Total} = EXP_{FI} + EXP_{SI}$$

Where:

EXP_{Total}	=	total exposure (*6.3 x 10^4 µg/kg bw/day*)
EXP_{FI}	=	exposure from soil invertebrate ingestion (*6.2 x 10^4 µg/kg bw/day*)
EXP_{SI}	=	exposure from ingestion of soil (*1183 µg/kg bw/day*)

Thus, the calculated PCB exposure rate from ingestion of both soil and soil invertebrates is 6.3 x 10^4 µg/kg bw/day.

As shrews are assumed to be the prey item for the ermine, and given that PCBs are known to bioaccumulate, a calculation was performed to estimate the steady-state PCB body burden in the shrew that results from the estimated PCB exposures obtained via the ingestion of soil and soil invertebrates. The calculated shrew body burden value is then fed to the ermine, assuming ermine consume shrew as whole organisms in general (which is common for predators given the small size of the shrew). This is a

relatively common ERA approach used when estimating food chain exposures to bioaccumulative substances and has been used (and accepted) on previous ERAs of PCB-impacted sites across Canada, including sites in Newfoundland and Labrador. Within this approach, it is assumed PCBs are accumulated for seven half-lives (at which point they reach an assumed equilibrium within the animal) over the animal's average lifespan. This is a conservative approach that likely overestimates PCB accumulation within prey items. It is also a highly uncertain approach in that there is much that is poorly understood about PCB bioaccumulation and elimination half lives in small wild mammals and the influence of metabolic rate and lipid content differences on PCB elimination kinetics. Limited validation of steady state body burden modelling for PCBs with measured PCB body burdens has been conducted at some locations within Canada, but the degree of agreement between modelled and measured results has been quite variable and mixed. Ideally, for sites where PCB uptake into wildlife is a concern, measured body burden data should be collected. Such data are not available in relation to the Site at this time.

Calculation of Shrew PCB Body Burden

$$BB_{Shrew} = \left[\left(\frac{EXP_{Total} \times BIO_{Oral}}{WB_{HL}} \right) \times \left(\frac{1 - e^{(-K2 \times DA \times WB_{HL})}}{1000} \right) \right] \div (1 - 0.68)$$

Where:

BB_{Shrew}	=	steady-state body burden in shrew (6.49 ug/g dry wt.)
EXP_{Total}	=	total exposure to the shrew (*6.3 x 10^4 µg/kg bw/day)*
BIO_{Oral}	=	oral bioavailability (*Assumed 1.0; unitless*)
WB_{HL}	=	whole body half-life (*Assumed 30 days; uncertainty is high as PCB elimination half-lives are highly variable across species, between different tissues and organs, and across PCB congeners; the assumed value is an estimate based on consideration of reported PCB half-lives in wild birds, mink and experimental animals (rats and mice) from WHO (1992), and also the short life span of the shrew and its high basal metabolic rate*.
K2	=	ln (2)/WB_{HL} (*$days^{-1}$*)(metabolic rate constant)
DA	=	duration of accumulation of chemical (*Assumed 7 half-lives*)
0.68	=	average fraction of water content of small mammals (Suter et al., 2000); used to convert wet weight body burden to a dry weight-based body burden.

Thus, the estimated shrew PCB body burden is estimated to be 6.49 µg/g dry wt.

Ermine PCB Exposures

Exposure via Shrew Consumption

$$EXP_{FI} = \frac{BB_{Shrew} \times FI_{Ermine} \times D_O}{BW_{Ermine}}$$

Where:

EXP_{FI}	=	exposure from food (shrew) ingestion (*686 µg/kg bw/day*)
BB_{Shrew}	=	steady-state PCB body burden in shrew (6.49 µg/g dry wt. as calculated above)
FI_{Ermine}	=	food intake rate (*9.41 g/day dry wt.)*
D_O	=	fraction of diet from Site (*Assumed 1.0; unitless*)
BW_{Ermine}	=	body weight (*0.089 kg*)

Thus, the calculated PCB exposure rate from ingestion of shrews is 686 µg/kg bw/day.

Exposure via Soil Ingestion

$$EXP_{Soil} = \frac{C_S \times SI_{Ermine} \times T_O}{BW_{Ermine}}$$

Where:

EXP_{Soil}	=	exposure from ingestion of soil (*80.2 µg/kg bw/day*)
C_s	=	concentration in soil (*27.1 µg/g*; *UCLM95)*
SI_{Ermine}	=	soil ingestion rate (*0.26 g/day)*
T_O	=	fraction of time spent on-site (*Assumed 1.0; unitless*)
BW_{Ermine}	=	body weight (*0.089 kg*)

Thus, the calculated PCB exposure rate from ingestion of soil is 80.2 µg/kg bw/day.

Total Exposure (from both soil and shrew ingestion)

$$EXP_{Total} = EXP_{FI_{Ermine}} + EXP_{SI_{Ermine}}$$

Where:

EXP_{Total}	=	total exposure (*766 µg/kg bw/day*)
$EXP_{FIErmine}$	=	exposure from food (shrew) ingestion (*686 µg/kg bw/day*)
$EXP_{SIErmine}$	=	exposure from ingestion of soil (*80.2 µg/kg bw/day*)

Thus, the calculated PCB exposure rate from ingestion of both soil and shrews is 766 µg/kg bw/day.

9.5 Effects Assessment

The effects assessment (also commonly referred to as the hazard or toxicity assessment) step of ERA evaluates the potential for chemical exposure to elicit an adverse effect or a toxic response in the ROCs. The toxicity of a chemical depends on the amount of chemical taken into an organism or its tissues and the duration of exposure (i.e., the length of time the receptor is exposed to the chemical). For every chemical, there is an exposure level or dose and duration of exposure necessary to produce a toxic effect in the ROCs (this is referred to as the exposure–response or dose-response relationship of a chemical). In the effects assessment, information relating to the exposure-response or dose-response relationships of each COPC is evaluated (usually from laboratory or captive animal studies) in order to determine an exposure or dose that is acceptable (unlikely to cause harm) in the ROCs selected for evaluation in the ERA. These values are commonly referred to as TRVs. Such values can exist for a number of endpoints

but the most commonly evaluated endpoints in ERA are effects on growth, reproduction and survival. This is consistent with the fact that for most ROCs evaluated in an ERA, the relevant level of biological organization is populations or communities. TRVs for a given COPC vary depending on the ROC under evaluation. The major outcome of the effects assessment step in an ERA is the identification of TRVs for each receptor-COPC combination that is assessed.

TRVs can be expressed in different ways depending on the COPC (and its properties) and the receptor or receptor group. Many TRVs are expressed as a dose (e.g., mg/kg body weight/day) and are commonly used to evaluate risks to mammalian, avian and herptile receptors via ingestion-based exposure pathways (and occasionally dermal and inhalation-based pathways). TRVs can also be expressed as environmental media or tissue concentrations (although few tissue-based TRVs have been developed to date). Such TRVs are often used in the ERA of receptors that are assessed as communities and in direct contact with an exposure medium (such as terrestrial vegetation and soil invertebrates in contact with soil). For these receptor groups, the TRVs used are generally regulatory soil quality benchmarks that considered these types of organisms in their derivation (e.g., CCME soil quality guidelines, U.S. EPA Ecological Soil Screening Levels (Eco-SSLs), Oak Ridge National Laboratory soil benchmarks, etc.).

It is important to recognize when/if ecological TRVs are exceeded by estimated exposures; it does not necessarily imply there is a risk of adverse ecological effects. Rather, it suggests further evaluation or consideration of additional LOE may be warranted before reaching final conclusions on the potential for ecological risk. This is discussed further in the risk characterization section below (Section 9.6).

For determining potential ecological risks from PCBs to the masked shrew and ermine, dose-based TRVs were reviewed from a number of North American sources of ecological TRVs. Table 9-5 presents the PCB TRVs that were selected for the shrew and ermine. It was not considered appropriate or necessary to apply uncertainty factors or any other toxicity extrapolation technique to the selected TRVs. To the extent possible, current guidance on ecological TRV best practices (i.e., Azimuth, 2012a,c; BC SAB, 2008; Allard et al., 2010) was followed with respect to selecting the TRVs for this ERA.

Table 9-5 Toxicity Reference Values (TRVs) for PCBs – Masked Shrew and Ermine

Receptor	TRV (µg/kg BW/day)	Basis of TRV	Endpoint(s)	Test Species	TRV Source
Masked Shrew	700	LOAEL for adverse reproductive effects in Oldfield mice orally exposed in diet to Aroclor 1254	Reproduction	Oldfield Mice	Sample et al., 1996
Ermine	700	LOAELs for adverse reproductive effects in mink orally exposed in diet to Aroclor 1242 and Aroclor 1254	Reproduction	Mink	Sample et al., 1996

Notes:

LOAEL = lowest observed adverse effect level.

9.6 Risk Characterization

In an ERA, the risk characterization step is the process by which the probability, magnitude and extent of adverse ecological effects (based on the information obtained from the exposure and effects assessments for each LOE) is integrated and interpreted in the context of the overall potential for ecological risk (Azimuth, 2012a). The risk characterization step also serves to translate the complex scientific information that comprises the previous steps of the ERA process into a format that is useful, unambiguous and understandable for risk managers. Another key element of risk characterization in an ERA is to acknowledge, evaluate and/or discuss the major strengths, limitations, conservative assumptions and uncertainties arising from the information used to estimate exposure and potential risk to the ROCs (Azimuth, 2012a; CCME, 1996).

The Azimuth (2012a) FCSAP ERA guidance indicates the following are typically key steps or considerations in risk characterization (depending on the scope of the ERA, as well as, the ROCs, chemicals, pathways and LOE evaluated):

- Relevance Check – Determine whether any deviations occurred during field, lab or modelling studies that could affect the relevance of the data for supporting LOEs. Adjustments to data supporting an LOE are made, if necessary. This can also include revisiting the site management and assessment goals, as well as, the assessment, measurement endpoints and LOE, to ensure they are appropriate and provide adequate linkages to each other.
- Interpret/Evaluate each LOE – Selection of appropriate methods to evaluate and interpret the information generated during the ERA for each LOE.
- Prepare Compiled Data Summary – A summary presentation of the data or outcomes for each LOE.
- Apply Weight of Evidence Procedure – Integrate the results of the various LOE using a WOE approach.
- Evaluate ERA Uncertainties – Consideration of the uncertainties, limitations, data gaps, and conservative assumptions that affect the interpretation of each LOE. Uncertainties must be evaluated in order to determine the level of confidence associated with risk estimates and LOEs, and to determine to what extent additional work may be warranted to reduce key areas of uncertainty and improve the accuracy of LOEs evaluated in the ERA.
- Consider Extrapolation / Interpolation – If necessary determine the degree to which ERA conclusions for some LOE can be expected to reliably translate to other site conditions or other LOE.
- Develop Site-Specific Benchmarks (if necessary) – Develop numerical values for COPCs in site media that may be used to distinguish action/no action levels.
- Summarize ERA Conclusions – Prepare a summary that characterizes ecological risk in terms of potential magnitude of response and other key attributes (such as likelihood, spatial extent, temporal extent, levels of biological organization that may be affected, causality, ecological relevance, etc.).

- Conduct Follow-Up Actions (if necessary) – Prepare recommendations (as necessary) for next steps in terms of corrective action, site closure, approvals, regulatory liaison, etc.

For the risk characterization of the shrew and ermine, a simple WOE approach was used. Azimuth (2012a) defines a WOE approach as "any process used to aggregate information from different lines of scientific evidence to render a conclusion regarding the probability and magnitude of harm". This definition encompasses a wide range of potential techniques and practices, which range from those that are qualitative and based on professional judgment to those that involve complex quantitative and/or statistical methods. The type of WOE approach used generally reflects the scale/scope, the level of effort and the numbers and types of LOE considered in the ERA. Irrespective of how the WOE approach is conducted, key principles are transparency, clarity, consistency and reasonableness (Azimuth, 2012a).

The specific WOE approach for risk characterization of the masked shrew and ermine was based on the following LOE:

- Comparing estimated PCB exposures to the TRV (i.e., ecological hazard quotients (EHQs)).
- Comparing the PCB Area spatial size and habitat quality to ASTM (2002) spatial criteria for ERA.
- Comparing the Site habitat quality to surrounding areas.
- Comparing receptor habitat preferences to Site habitat conditions and habitat quality.
- Interpreting Ns values with respect to likelihood for populations or individuals to use or occur on the Site.

Prior to ecological risk characterization, a relevance check was conducted between the ERA objectives and goals with respect to the shrew and ermine, and it was determined that the selected assessment and measurement endpoints and LOE for these receptor groups were consistent with the ERA goals and objectives.

With respect to EHQs, the calculation consists of a simple ratio between the estimated exposure rate for a given ROC and the applicable TRV, as follows.

$$\textit{Ecological Hazard Quotient (EHQ)} = \frac{\textit{Estimated Exposure (µg/kg BW/day)}}{\textit{TRV(µg/kg BW/day)}}$$

It is standard ERA practice to use a target EHQ value of 1.0. Thus, if the calculated EHQ is less than 1.0, exposures are lower than the TRV and it is typically concluded that the potential for adverse effects is low or negligible. However, if the calculated EHQ exceeds 1.0, meaning exposure is greater than the TRV, it does not necessarily indicate that adverse effects are likely. Rather, the assumptions and data used in the ERA, for all LOE, are reviewed prior to determining whether or not there is a potential for ecological risk for a particular ROC, and if further assessment appears warranted. Consideration of the key uncertainties,

limitations and conservative assumptions within the ERA are also important factors in ecological risk characterization.

It has become relatively common among ERA practitioners in recent years to consider traditional quantitative ERA modeling outcomes (i.e., EHQs) as a means to rule out certain chemicals, receptors and exposure pathways from further evaluation, rather than relying on such outcomes as definitive or representative estimates of potential ecological risk. The current FCSAP ERA guidance (Azimuth, 2012a) notes that HQs are simple ratios, and that situations where an EHQ exceeds 1.0 only indicates an adverse response is possible, and that more precise or accurate evaluation of ecological risks may be warranted to address uncertainty. This guidance further notes where EHQs are calculated, care must be taken not to infer more information from the ratio than is warranted. While EHQs are relatively easy to derive, they are often misinterpreted (Allard et al. 2010) with common errors including the belief that an EHQ is directly proportional to the magnitude of risk. EHQs neither contain information about the specific probability that an adverse effect will occur nor do they convey any information about the magnitude of a potential adverse effect (Azimuth, 2012a). The FCSAP ERA guidance further elaborates on key items that must be considered to put EHQs in perspective and use them as meaningful LOE in ERAs.

9.7 Ecological Risk Assessment Results

9.7.1 Current Scenario (As Is – Outdoor Site Visitor (Trespasser))

This section summarizes the results for each LOE considered in the ERA.

EHQ calculations for the masked shrew and ermine are presented below.

Masked Shrew Ecological Hazard Quotient

$$EHQ = \frac{EXP_{Total}}{TRV}$$

Where:

EHQ	=	ecological hazard quotient (89.7*; unitless*)
EXP_{Total}	=	total exposure (*6.3 x 10^4 µg/kg bw/day*; See Section 5.4.2)
TRV	=	toxicity reference value (700 *µg/kg/day*)

Ermine Ecological Hazard Quotient

$$EHQ = \frac{EXP_{Total}}{TRV}$$

Where:

EHQ	=	ecological hazard quotient (1.1; *unitless*)
EXP_{Total}	=	total exposure (766 *µg/kg bw/day*; See Section 5.4.2)
TRV	=	toxicity reference value (700 *µg/kg/day*)

The calculated EHQ (89.7) for the masked shrew suggests a moderate to high potential for adverse effects under the exposure conditions evaluated in the ERA. For the ermine, the calculated EHQ (1.1) only slightly exceeds 1.0, which suggests a low potential for ecological risk. The EHQ values are specific to these receptors, exposure pathways, parameters and various conservative assumptions used to generate these values. As described previously, and summarized below in Section 9.8, the approaches and assumptions used to calculate EHQs for the masked shrew and ermine are considered to be highly conservative overall and likely substantially overestimate the potential for ecological risk in these ROCs.

As previously noted in Section 9.6, while EHQs can indicate if the assessed exposure conditions pose a potential for ecological risks, they are not definitive or even accurate representations of true ecological risk, and cannot predict the specific probability that an adverse effect will occur in a given ROC nor convey information about the possible magnitude of a potential adverse effect. As such, it has become common in ERAs to rely equally or more on other LOE relative to EHQ values, when using an overall WOE approach to determine if the potential for ecological risk is significant enough and reliable enough to merit further study or corrective action.

Outcomes for the other LOE considered in the ERA are summarized in the following bullets (presented in the same order as noted in Section 9.6).

- The PCB Area has a total area of approximately 913 m^2 (or 0.091 ha) (AMEC, 2013). The total areal extent of the PCB Area is less than the ASTM (2002) spatial criterion of 0.81 ha. In addition, not all portions of the PCB Area have soil PCB impacts, thus the area available for potential PCB exposure to ecological receptors is actually less than 0.091 ha.
- When the Site habitat quality is considered relative to habitat quality on adjacent or surrounding sites/areas, it is evident that virtually all of the areas surrounding the Site offer better habitat suitability for small mammals and birds than the Site does. There is nothing to distinguish the Site from these other areas as offering unique or preferred habitat for any ecological receptor population or community.
- Review of habitat preferences for the masked shrew and ermine (from information provided in: Azimuth, 2012a,b; U.S. EPA, 1993; Arenal et al., 2006; and, Natureserve Explorer (http://www.natureserve.org/explorer), indicates Site conditions are not consistent with the preferred habitat types for these ROCs (and other small mammals and birds). Thus, it can be concluded the Site does not offer significant suitable habitat conditions for common small mammal and avian receptors. While this does not mean that individual animals (out of these receptor groups) will not periodically occur on, or use the Site, the probability that local populations of these receptors would use the Site (for foraging, nesting, breeding, etc.) is considered to be very low.
- Calculated values for N_s (the number of individuals of a given receptor species that are likely to inhabit any habitat subdivision on a site) for the masked shrew, ermine and a few other commonly assessed ROCs in terrestrial ERAs, suggest that few (if any) individual organisms (of the species

considered) would be expected to occur on or use the Site to any significant extent. As previously described (Section 9.3.2), N_s is calculated using conservative estimates for ROC home/foraging range size and the spatial area of the Site (PCB Area). While the N_s calculation is not without its uncertainty, it provides a reasonable indication of the numbers of individual organisms that may be expected to use a habitat subdivision on a site. The small numbers obtained for N_s (See Table 9-2) are clearly not indicative of populations of these receptors.

Collectively, these LOE suggest the Site is unlikely to provide habitat and food resources that can support local populations of small mammalian or avian receptors and the use of the Site by such receptors (for foraging, nesting, breeding, resting, etc.) would likely be very limited.

9.7.2 Future Scenario (Post-Capping)

Based on the fact that an engineered cap will be placed over the PCB Area, a secondary ERA exposure scenario was evaluated that determines if the cap is likely to minimize potential ecological exposures and risks. As in the HHRA, this scenario is evaluated qualitatively based on such considerations as depth of the cap layers, the types of cap layers, the outcomes of the primary Current Conditions ERA scenario, and the review of environmental media and biota chemistry data that has been collected to date in relation to the Site.

Although the final design of the engineered cap has not been completed, the cap will be approximately 2.0 m thick and consist of imported clean fill, bedding material, a geomembrane liner and a vegetative layer. The top layer will be graded to prevent ponding of water.

This barrier will prevent direct contact between PCB-impacted soils and any mammalian or avian ecological receptors that may occur or forage on the Site. The cap will negate direct soil contact pathways for all ecological receptors to residual PCB concentrations in what is currently surface and subsurface soil within the PCB Area.

While the Site conditions post-capping may be more amenable (relative to current conditions) to ecological receptors as potential habitat, or a potential food resource/foraging area, all PCB-impacted soils within the PCB Area will not be accessible, as these soils will be buried under approximately 2.0 m of engineered cap materials. As the most relevant soil depth for ecological receptors is considered to be within 30 cm of the soil surface (Anderson et al., 2010; Suter II, 2007; Suter et al., 2000), there is a negligible potential that receptors could come into direct contact with residual PCB impacted soil, after the cap is in place.

In conclusion, the engineered cap is expected to fully mitigate potential ecological exposures that were estimated in the Current Scenario (As Is – Outdoor Site Visitor (Trespasser)). If there is no or negligible exposure, then there is no or negligible potential for ecological risk. Furthermore, there is no evidence to

date that PCBs are migrating from the PCB Area to off-site terrestrial or aquatic media or biota, based on the groundwater, surface water, sediment and fish monitoring programs that have been conducted since 2003. With the exception of fish, these monitoring events have now occurred over multiple sampling events in multiple years and have consistently indicated PCB concentrations below the laboratory RDL. The engineered cap will further reduce the potential for PCBs to migrate off-site by removing any potential surface media transport pathways such as wind erosion of impacted soil and bulk transport of impacted soil via surface runoff.

9.8 Uncertainty, Variability, Limitations and Conservative Assumptions in the ERA

ERA involves assigning numerical values to various parameters in order to obtain estimates of exposure and risk. Variability and uncertainty in these values leads to variability and uncertainty in the estimates of exposure and risk. The conclusions of any ERA are dependent on the data and assumptions that are evaluated within it, and thus, are greatly influenced by the variability and uncertainty associated with the data and assumptions. It is therefore important in an ERA to characterize the key areas of variability and uncertainty (and any other major study limitations) to avoid possibly underestimating exposures and risks, to the extent possible, and to recognize when exposures and risks have likely been substantially overestimated. An evaluation of uncertainty and variability provides information that helps risk managers make appropriate decisions regarding whether or not risks need to be managed, how the risks can best be managed, and can identify situations where the use of more sophisticated approaches and/or further data collection can reduce or refine key sources of uncertainty and/or variability.

Uncertainty should not be confused with variability. Uncertainty is a lack of confidence in a result or estimate stemming from limited data or missing information. Variability describes differences in parameter values such as chemical concentrations at different locations on a site, differences in body weight or food intake rates for individual animals. In other words, variability is defined by the range or "spread" of values in a given population, which is influenced by sample size, repeated measures and area of coverage.

The inherent tendency of ERAs to overestimate exposures and risks to ecological receptors favours Type I errors (false positives) and reduces the probability of Type II errors (false negatives). In the evaluation of uncertainty and variability, what is ultimately most important is confidence that one has not under-predicted exposures and risks and that the approaches and assumptions used in the ERA will avoid or reduce the occurrence of Type II errors.

A key question that is often raised when characterizing uncertainty and variability is: "Will the collection of more data improve the understanding of the variability and/or reduce uncertainty?" At some point, the collection of additional data will reach the point of diminishing returns, when the effort and resources that

are expended to further understand variability and reduce uncertainty are no longer producing meaningful improvements. For example, if additional soil sampling/analysis was conducted, and the new data yielded concentrations that fell well within the range of existing data, with no substantial changes to values that measure the "spread" of the data (such as variance, standard error, standard deviation, coefficient of variation etc.), then the additional sampling does not provide any added value.

Where variability and/or uncertainty are known to exist, it is standard ERA practice to make assumptions and select data that overestimate, rather than underestimate potential exposure and risk. In general, given most ERAs make conservative assumptions when faced with uncertainty and variability, potential exposures and risks tend to be overestimated (often considerably) for the receptors, exposure pathways and chemicals evaluated.

The following bullets describe the major sources and areas of uncertainty and variability in the ERA. Overall, given the outcomes of the ERA (and the conservative assumptions within it) and considering the Site's small size, habitat quality and future land use, there is a high degree of confidence that appropriate receptors, exposure pathways and chemicals have been evaluated, and that exposures and risks have not been underestimated.

- PCB surface soil concentration data were considered to be represented by samples from depths down to 1.5 m below ground surface (as per CCME subsurface/surface soil cutoff depth). While it is common for relevant soil depths in an ERA to vary depending on the ROC, the source of contamination and site conditions, soil depths of approximately 0 to 25 or 30 cm are generally considered most reasonable for ERA purposes (Anderson et al., 2010; Suter II, 2007; Suter et al., 2000). As the PCB-containing transformers were buried at the PCB Area, as opposed to being left at ground surface, it is likely that the top 30 cm of the soil profile is less impacted than soil at greater depths. Thus, the use of soil data from depths down to 1.5 m below surface likely overestimates potential PCB exposures to the masked shrew and ermine in the ERA, as it was assumed that these receptors could potentially come into contact with measured surface soil concentrations of COPCs.
- The ERA assumed the masked shrews forage entirely in areas of the Site that contain PCB-impacted soils and that the shrew diet consists entirely of soil invertebrates from the Site. The ERA also assumed the ermine diet is comprised entirely of shrews that forage in areas of the Site that contain PCB-impacted soils and that soil exposure incurred by the ermine happens in areas of the Site that contain PCB-impacted soils. While these are typical assumptions for an ERA of a small contaminated site, they are considered to be highly unrealistic and conservative.
- The approach used to estimate the masked shrew PCB body burden (which was then fed to the ermine in the ERA) is a relatively common ERA approach used when estimating food chain exposures to bioaccumulative substances and has been used (and accepted) on previous ERAs of PCB-impacted sites across Canada, including sites in Newfoundland and Labrador. While it is believed to be inherently conservative, there are a number of assumptions and uncertainties within this approach that

are not well validated with measured small mammal PCB body burden data (including the assumption that PCB body burdens are at equilibrium after 7 half-lives have elapsed over the animal's average lifespan). The lifespan of a shrew is highly variable and usually on the order of a few to several months. Short lifespans would limit the amount of PCBs that may be accumulated by shrews. In addition, there is much that is poorly understood about PCB bioaccumulation and elimination half lives in small wild mammals and the influence of metabolic rate and lipid content differences on PCB elimination kinetics. Only limited validation of steady state body burden modelling for PCBs with measured PCB body burdens has been conducted at some locations within Canada, with the degree of agreement between modelled and measured results being quite variable and mixed. Ideally, for sites where PCB uptake into wildlife is a concern, measured body burden data should be collected. Such data has not been collected to date.

- Receptor body weights and other key physiological and behavioural parameters were obtained from reliable regulatory agency guidance documents or scientific literature sources. There is some uncertainty associated with these values though, as they are not site-specific or necessarily representative of what occurs within the local receptor populations.
- Data on wildlife food ingestion rates are only available for a few species, primarily due to the difficulties in measuring such intakes for free-ranging wildlife. As such, for specific receptors (such as the masked shrew and ermine), it is often necessary to use allometric equations to estimate food ingestion rates for ROCs. Allometric equations assume food intake is proportional to body weight, which may not necessarily be the case.
- Published soil ingestion rates do not exist for many mammalian and avian receptors. Thus, it is common ERA practice to assume (based on literature and/or regulatory guidance) that a certain percentage of the receptor's overall food ingestion rate represents a given receptors' soil ingestion rate.
- In any ERA, it is inherently difficult to assign representative diets with fixed proportions of dietary items to the assessed ROCs. For any ROC, even those with a narrow range of dietary preferences, diets can be highly variable and difficult to estimate with accuracy (for example, the proportion of dietary items for a ROC may vary between locations or individuals, and seasonally). ERAs typically account for this uncertainty by making conservative assumptions about receptor diets such that worst-case diets are frequently assumed for the ROCs evaluated in an ERA. This was the case in this ERA as well, wherein it was assumed that shrews consume 100% soil invertebrates from the Site, and ermine diets consist of 100% of shrews that forage on the Site.
- The oral bioavailability of PCBs in food items and soil was conservatively assumed to be 100% (or, the same as in the toxicology studies that the TRVs were derived from). This assumption likely overestimates PCB exposure to the shrew and ermine as the gastrointestinal absorption of most chemicals from complex matrices such as foods and soil are rarely 100%, and can be quite low, depending on various physical and chemical properties of the food items and soil. Toxicity data directly related to the ROCs are often unavailable or limited. Therefore, many of the TRVs used in an ERA are derived from similar or related species exposed to the COPCs under controlled laboratory

conditions that are designed to maximize the potential for measurable adverse effects. Extrapolation of laboratory toxicity data to other species may involve the use of uncertainty factors, depending on how similar (or not) the test species and the ROC species are (in terms of taxonomy, gastrointestinal physiology, feeding guild, etc.). The TRVs used in this ERA were selected from standard sources of wildlife TRVs and were considered reasonably applicable to the shrew and ermine such that toxicity extrapolation approaches were not necessary.

- While the selection of COCs is believed to be reasonable and appropriate for the Site, there is always some possibility that there are species living in (or possibly extirpated from) the Site that may be more sensitive to one or more of the site-related chemicals than those receptors that were evaluated in the ERA.
- Treatment of data for the ERA was conducted in a manner that is intentionally conservative. This approach was taken to minimize the potential of exposures and risks associated with PCBs present in Site soil to be underestimated. For example, concentrations of PCBs in soil that were below the laboratory RDL were assumed to be present at the RDL, and the higher of the field/lab duplicate or original sample results were selected for evaluation in the ERA. This approach is likely to overestimate potential PCB exposures and risks.
- This ERA only assesses chemical stressors and only PCBs. Other chemical stressors as well as common non-chemical ecological stressors such as predation, disease, habitat loss/fragmentation and competition are not evaluated in either a quantitative or qualitative manner. It is not known if other chemicals present in the PCB Area soils are more or less likely to result in adverse ecological effects than PCBs. Also, on any given site, non-chemical stressors may interact with chemical stressors in complex ways and can often be of greater biological or ecological significance than the presence of chemical contaminants in site media. For the Site, key non-chemical stressors would likely include predation, human disturbance from recreational use of the Site and adjacent areas, and habitat fragmentation.
- As in the HHRA, the use of deterministic (or point estimate) exposure analysis techniques in the ERA tends to overestimate potential exposures and risks. In this ERA, UCLM95 soil concentrations of PCBs were evaluated as the EPCs.
- The ERA assumed that there is no attenuation, transformation, degradation or loss of PCBs in soils.
- While the lack of detectable PCB concentrations in raw fish fillet samples (skin and bones removed; collected in 2006) suggests PCBs have not migrated to local water bodies, it must be recognized that fillets (muscle tissue) are not necessarily the most appropriate manner in which to prepare and analyze fish tissues when the concern is related to PCB exposure via fish consumption (by piscivores). It is known that PCBs accumulate preferentially in fatty tissues and organs and that piscivores tend to consume whole fish more so than selected tissues from fish prey species. Thus, the analysis of raw fillets may not have captured potential PCB contamination (if present) in local whole prey-sized fish. However, PCBs were not detected in the historical surface water samples, and as such, PCBs in fish (i.e., associated with the PCB Area) would not be anticipated.

- All PCB media and biota chemistry data collected to date are expressed as total PCBs (with Aroclor resemblance also reported for samples with detectable PCB concentrations). Data has not been collected to date for specific PCB congeners, including the "dioxin-like" coplanar PCB congeners. Given that burning occurred in some areas of the WDS (although, apparently not within the PCB Area), and that dioxins and furans have been measured in environmental media of the WDS, consideration could be given to conducting PCB congener analysis in a future monitoring event to determine if there are elevated media and/or biota concentrations of coplanar PCB congeners that would be expected to contribute to potential effects related to the co-presence of dioxin and furan congeners. At this time, it is not possible to evaluate the potential contribution of coplanar PCBs to the effects that may be caused by the presence of dioxins and furans in the WDS media.

10.0 CONCLUSIONS

10.1 HHRA Conclusions

Based on the HHRA results presented in Sections 8.4.1, and 8.4.2, and considering the major uncertainties, limitations and conservative assumptions that relate to the HHRA (Section 8.6), the following conclusions are made:

- PCB concentrations in soil are not considered to pose a human health concern for the soil to outdoor air exposure pathway.
- In the Current Scenario (As Is - Outdoor Site Visitor (Trespasser)), consideration of the HHRA outcomes, as well as the uncertainty related to the current carcinogenic status of PCBs in Canada, suggests the need to consider corrective action in relation to soil PCB concentrations at the Site, despite the high likelihood that the conservative assumptions used within the HHRA have substantially overestimated potential exposures and risks. This is in keeping with common site management practices for PCBs (where some corrective action is often taken irrespective of human health and/or ecological risk potential) and is also in keeping with the fact that PCBs are frequently associated with a high potential for public concern.
- When it is assumed that PCBs are human carcinogens and cancer-based TRVs are used, higher human health risk estimates are generated in the Current Scenario (As Is - Outdoor Site Visitor (Trespasser)), relative to assuming PCBs are not human carcinogens. However, concerns regarding cancer development in HHRAs are primarily associated with situations of chronic or continuous exposure to a given carcinogen(s), where there is a high frequency and duration of exposure to carcinogenic substances. This is not the case for the Site where human exposure to PCB-impacted soils would occur on a transient or intermittent basis and for short durations when it occurs. Such exposure conditions are generally not associated with a high potential for developing cancer, and there are no known studies to date (for any carcinogen) that have been able to demonstrate a strong

link between cancer development and short term, low frequency, low duration exposures to the generally low levels of carcinogenic chemicals that may occur in ambient environmental media.

- The HHRA is considered sufficiently conservative and protective such that the overall potential for human health risk is low and the PCB Area does not pose any immediate human health concern.
- Soil SSTLs were not developed as part of this HHRA given consideration of the HHRA outcomes and major uncertainties, and most importantly, considering the WDS closure activities will involve corrective action for PCB-impacted soil. Specifically, the PCB Area will be covered with an engineered cap.
- The evaluation of the Future Scenario (Post-Capping) in the HHRA indicates the engineered cap is expected to fully mitigate potential human exposures that were estimated in the Current Scenario (As Is - Outdoor Site Visitor (Trespasser)). If there is no or negligible exposure, then there is no or negligible potential for human health risks. It is understood that the GNLDEC are planning to install the engineered cap over the PCB Area in the Fall of 2013.
- There is also no evidence to date that PCBs are migrating from the PCB Area to off-site terrestrial or aquatic media or biota, based on the previous groundwater, surface water, sediment and fish monitoring programs that have been conducted since 2003. With the exception of fish, these monitoring events have occurred over multiple sampling events in multiple years and have consistently indicated PCB concentrations below the laboratory RDL. The engineered cap will further reduce the potential for PCBs to migrate off-site by removing any potential surface media transport pathways such as wind erosion of impacted soil and bulk transport of impacted soil via surface runoff.

10.2 ERA Conclusions

Based on the ERA results for each LOE that was evaluated (Section 9.7), and considering the major uncertainties, limitations and conservative assumptions that relate to the ERA (Section 9.8), it is concluded that the potential for ecological risk is low for the masked shrew and the ermine, in relation to the Site. As these ROCs would be expected to incur a greater degree of potential PCB exposure from the Site than other small mammals and birds, it follows that the potential for ecological risk in other mammalian and avian receptors would also be low and is likely negligible. The LOE evaluated in the ERA suggest a low likelihood for even individual shrews or ermine to occur on or use the Site. While the possibility exists that individual animals could incur PCB exposure from the Site on an occasional basis, it is extremely unlikely that such exposures would affect local populations of either masked shrew or ermine. Given these findings, there is no apparent need for mitigation of the PCB Area with respect to potential ecological risks. Support for these conclusions is provided in the following bullets.

- While the calculated EHQ values for the masked shrew and ermine suggest a moderate to high potential for adverse effects and a low potential for adverse effects, respectively, the approaches and assumptions used to calculate these EHQs are considered to be highly conservative and likely substantially overestimate the potential for ecological risk in these ROCs.

- As a LOE, EHQs are limited in that they are not definitive or even accurate representations of true ecological risk and cannot predict the specific probability that an adverse effect will occur in a given ROC nor convey information about the possible magnitude of a potential adverse effect. As such, it has become common in ERAs to rely equally or more on other LOE relative to EHQ values, when using an overall WOE approach to determine if potential ecological risks exist.
- The LOEs related to spatial size of the PCB Area, the Site's habitat quality relative to habitat quality on adjacent or surrounding sites/areas, habitat preferences for the selected ROCs (masked shrew and ermine) and the calculated N_s values collectively suggest the Site is unlikely to provide habitat and food resources that can support local populations of small mammalian or avian receptors, and the use of the Site by such receptors (for foraging, nesting, breeding, resting, etc.) would likely be very limited.

Soil SSTLs were not developed as part of this ERA, given the conclusion of a low potential for ecological risk in the masked shrew and ermine, under a set of conservative assumptions and approaches that overestimate exposure and risk. Another and more important reason for not developing soil SSTLs is the fact that the WDS closure activities will involve corrective action for PCB-impacted soils. Specifically, the PCB Area will be covered with an engineered cap.

While Site conditions post-capping may be more amenable (relative to current conditions) to ecological receptors as potential habitat, or a potential food resource/foraging area, all PCB-impacted soils within the PCB Area will not be accessible, as these soils will be buried under approximately 2.0 m of engineered cap materials. As the most relevant soil depth for ecological receptors is considered to be within 30 cm of the soil surface (Anderson et al., 2010; Suter II, 2007; Suter et al., 2000), there is a negligible potential that receptors could come into direct contact with residual PCBs in soil, after the cap is in place.

The evaluation of the Future Scenario (Post-Capping) in the ERA indicates the engineered cap is expected to fully mitigate potential ecological exposures that were estimated in the Current Scenario (As Is - Outdoor Site Visitor (Trespasser)). If there is no or negligible exposure, then there is no or negligible potential for ecological risk. It is understood that the GNLDEC are planning to install the engineered cap over the PCB Area in the Fall of 2013.

Furthermore, there is no evidence to date that PCBs are migrating from the PCB Area to off-site terrestrial or aquatic media or biota, based on the previous groundwater, surface water, sediment and fish monitoring programs that have been conducted since 2003. With the exception of fish, these monitoring events have now occurred over multiple sampling events in multiple years and have consistently indicated PCB concentrations below the laboratory RDL. The engineered cap will further reduce the potential for PCBs to migrate off-site by removing any potential surface media transport pathways such as wind erosion of impacted soil and bulk transport of impacted soil via surface runoff.

11.0 RECOMMENDATIONS

Based on data collected to date for the PCB Area, and the outcomes and conclusions of the HHRA and ERA, there does not appear to be a need for further site assessment or risk assessment studies for the PCB Area. The planned capping of the PCB Area is considered a sufficient degree of mitigation that will prevent future exposures for both human and wildlife receptors to residual PCBs present in the PCB Area soil.

However, once the closure planning process for the WDS has been completed, and the engineered cap has been installed, there will be a need for continued environmental monitoring, to assess and maintain the integrity of the cap and leachate collection system.

It is understood that groundwater and surface water will continue to be monitored. If PCB impacts are identified in these media, additional soil and sediment sampling/analysis will be conducted at that time. Once the cap has been installed over the PCB Area, there will not be a need for any further soil sampling activities in this area.

12.0 CLOSURE

This report was prepared exclusively for the purposes, project, and site location(s) outlined in the report. The report is based on information provided to, or obtained by Dillon Consulting Limited ("Dillon") as indicated in the report, and applies solely to site conditions existing at the time of the site investigation(s). Although a reasonable investigation was conducted by Dillon, Dillon's investigation was by no means exhaustive and cannot be construed as a certification of the absence of any contaminants from the site(s). Rather, Dillon's report represents a reasonable review of available information within an agreed work scope, schedule and budget. It is therefore possible that currently unrecognized contamination or potentially hazardous materials may exist at the site(s), and that the levels of contamination or hazardous materials may vary across the site(s). Further review and updating of the report may be required as local and site conditions, and the regulatory and planning frameworks, change over time.

This report was prepared by Dillon for the sole benefit of the Government of Newfoundland and Labrador Department of Environment and Conservation. The material in it reflects Dillon's best judgment in light of the information available to it at the time of preparation. Any use which a third party makes of this report, or any reliance on or decisions made based on it, are the responsibilities of such third parties. Dillon accepts no responsibility for damages, if any, suffered by any third party as a result of decisions made or actions based on this report.

This report was prepared by Randy Fancey, CET, and Rob Willis, B.Sc., M.E.S., EP, and reviewed by Andrew J. Blackmer, M.Sc., P.Geo.

Andrew J. Blackmer, M.Sc., P.Geo.
Site Professional

Rob Willis, B.Sc., M.E.S., EP
Senior Toxicologist and Risk Assessor

13.0 REFERENCES

Allard, P., A. Fairbrother, B. Hope, R.N. Hull, M.S. Johnson, L. Kapustka, G. Mann, B. McDonald, B.E. Sample. 2010. Recommendations for the development and application of wildlife toxicity reference values. Integrated Environmental Assessment and Management. 6:28-37.

AMEC. 2013. Letter Report to Ms. Christa Curnew, P.Eng., Department of Environment & Conservation. Re: Test Pitting and Soil Sampling Program, PCB Disposal Area, Upper Trinity South (New Harbour), Waste Disposal Site at New Harbour Barrens, NL. AMEC Project No: TF1312736.

AMEC. 2011. Final Closure Plan. Upper Trinity South (New Harbour) Waste Disposal Site. Submitted To: Department of Environment and Conservation. Submitted By: AMEC Earth & Environmental. March, 2011. AMEC Project No. TF1112731.

AMEC. 2010. Data cited in: SNC, 2012.

AMEC. 2008. Groundwater and Surface Water Monitoring Program at the Upper Trinity South (New Harbour) Waste Disposal Site. Submitted To: Department of Environment and Conservation. Submitted By: AMEC Earth & Environmental. March, 2008. AMEC Project No. TF7112716.

AMEC. 2007a. Upper Trinity South (New Harbour) Waste Disposal Site 2006 Fish Sampling Program. Submitted to: Department of Environment and Conservation. Submitted by: AMEC Earth & Environmental. May 11, 2007.

AMEC. 2007b. Upper Trinity South (New Harbour) Waste Disposal Site Implementation of Leachate Control System. Submitted to: Department of Environment and Conservation. Submitted by: AMEC Earth & Environmental. March, 2007. AMEC Project No. TF6112714.

AMEC. 2006. Upper Trinity South (New Harbour) Waste Disposal Site Design of Leachate Control System. Submitted to: Department of Environment and Conservation. Submitted by: AMEC Earth & Environmental. June, 2006. AMEC Project No. TF5112710.

Anderson, R.H., Prues, A.G., and Kravitz, M.J. 2010. Determination of the biologically relevant sampling depth for terrestrial ecological risk assessments. Geoderma. 154(3-4):336-339.

Arenal, C.A., Sample, B.E., Ziolowski, D.J., and Johnson, M.S. 2006. Development of Terrestrial Exposure and Bioaccumulation Information for Ecological Risk Assessments. Contract No. DAAD050-00-P-8365. U.S. Army Center for Health Promotion and Preventive Medicine, Toxicology Directorate, Health Effects Research Program.

Arnold, D.L., F. Bryce, R. Stapley et al. 1993a. Toxicological consequences of Aroclor 1254 ingestion by female Rhesus (Macaca mulatta) monkeys, Part 1A: Prebreeding phase - clinical health findings. Food Chem Toxicol. 31:799- 810.

Arnold, D.L., F. Bryce, K. Karpinski et al. 1993b. Toxicological consequences of Aroclor 1254 ingestion by female Rhesus (Macaca mulatta) monkeys, Part 1B: Prebreeding phase -clinical and analytical laboratory findings. Food Chem Toxicol. 31:811-824.

ASTM. 2011. Designation: E2385 – 11. Standard Guide for Estimating Wildlife Exposure Using Measures of Habitat Quality.

ASTM. 2002. Designation: E 2205 – 02. Standard Guide for Risk-Based Corrective Action for Protection of Ecological Resources.

ATSDR. 2004a. Guidance Manual for the Assessment of Joint Toxic Action of Chemical Mixtures. U.S. Department of Health and Human Services, Agency for Toxic Substances and Disease Registry. http://www.atsdr.cdc.gov/interactionprofiles/ipga.html.

ATSDR. 2004b. Interaction profile for Persistent Chemicals found in Fish (chlorinated dibenzo-p-dioxins, hexachlorobenzene, p,p'-DDE, methylmercury, PCBs. Atlanta, GA: U.S. Department of Health and Human Services, Public Health Service, Agency for Toxic Substances and Disease Registry. http://www.atsdr.cdc.gov/interactionprofiles/ip01.html.

ATSDR. 2004c. Interaction profile for persistent chemicals found in breast milk (chlorinated dibenzo-p-dioxins, hexachlorobenzene, p,p'-DDE, methylmercury, PCBs. Atlanta, GA: U.S. Department of Health and Human Services, Public Health Service, Agency for Toxic Substances and Disease Registry. http://www.atsdr.cdc.gov/interactionprofiles/ip03.html.

ATSDR. 2004d. Interaction profile for Cesium, Cobalt, Polychlorinated Biphenyls, Strontium, and Trichloroethylene, Atlanta, GA: U.S. Department of Health and Human Services, Public Health Service, Agency for Toxic Substances and Disease Registry. http://www.atsdr.cdc.gov/interactionprofiles/ip07.html.

ATSDR. 2000. Toxicological Profile for Polychlorinated Biphenyls (PCBs). Agency for Toxic Substances and Disease Registry (ATSDR), U.S. Department of Health and Human Services, Public Health Service, Agency for Toxic Substances and Disease Registry. Atlanta, GA. http://www.atsdr.cdc.gov/ToxProfiles/tp.asp?id=142&tid=26.

Azimuth. 2012a. Federal Contaminated Sites Action Plan (FCSAP) Ecological Risk Assessment Guidance. Prepared for: Environment Canada. Prepared by: Azimuth Consulting Group.

Azimuth. 2012b. FINAL. Federal Contaminated Sites Action Plan (FCSAP) Ecological Risk Assessment Guidance. Module C: Standardization of Wildlife Receptor Characteristics. Prepared for: Environment Canada. Prepared by: Azimuth Consulting Group.

Azimuth. 2012c. FCSAP Supplemental Guidance for Ecological Risk Assessment. Selection or Development of Site-specific Toxicity Reference Values. Prepared for: Environment Canada. Prepared by: Azimuth Consulting Group.

Baars, A.J. et al. 2001. Re-evaluation of human-toxicological maximum permissible risk levels. RIVM report no. 711701025. National Institute of Public Health and the Environment, Bilthoven, The Netherlands, March 2001. Available at: http://www.rivm.nl/bibliotheek/rapporten/711701025.pdf.

Barnthouse, L.W. and Brown, J. 1994. Conceptual model development. Chapter 3 in Ecological Risk Assessment Issue Papers. EPA/630/R-94/009. U.S. Environmental Protection Agency, Washington, DC. Cited In: Suter II, 2007.

BC SAB. 2008. Report on: Detailed Ecological Risk Assessment (DERA) in British Columbia. Technical Guidance. Submitted to: BC Ministry of Environment. Submitted by: Science Advisory Board for Contaminated Sites in British Columbia.

Bowman, R.E., Heiromimus, M.P., and Barsotti, D.A. 1981. Locomotor hyperactivity in PCB-exposed rhesus monkeys. Neurotoxicol. 2(2):251–268.

Casteel, S.W., Weis, C.P., Henningsen, G.M., and Brattin, W.J. 2006. Estimation of Relative Bioavailability of Lead in Soil and Soil-Like Materials Using Young Swine. Environ. Health Perspect 114:1162–1171.

CCME. 2006. A Protocol for the Derivation of Environmental and Human Health Soil Quality Guidelines. Canadian Council of Ministers of the Environment. PN 1332. ISBN 10-896997-45-7.

CCME. 1996. A Framework for Ecological Risk Assessment: General Guidance. Canadian Council of Ministers of the Environment. CCME Subcommittee on Environmental Quality Criteria for Contaminated Sites. The National Contaminated Sites Remediation Program, Winnipeg, MB.

EAUK. 2005. Environment Agency's Science Update on the use of Bioaccessibility Testing in Risk Assessment of Land Contamination. February, 2005. Environment Agency U.K. Available at: http://www.environmentagency.gov.uk/commondata/acrobat/science_ update_1284046.pdf.

Environment Canada. 2001. Canadian Soil Quality Guidelines for Polychlorinated Biphenyl (PCBs): Environmental Health. Report No. 1-2. Scientific Supporting Document. National Guidelines and Standards Office, Environmental Quality Branch, Environment Canada. En40-236/1-2001E.

Geology of the Island of Newfoundland, Geological Survey Branch, Department of Mines and Energy,, Government of Newfoundland and Labrador, Map 90-01, 1990.

Gron, C., and Andersen, L. 2003. Human bioaccessibility of heavy metals and PAH from soil. Environmental Project No. 840, Technology Programme for Soil and Groundwater Contamination. Danish Environmental Protection Agency, Danish Ministry of the Environment.

Health Canada. 2010a. Federal Contaminated Site Risk Assessment in Canada Part I: Guidance on Human Health Preliminary Quantitative Risk Assessment (PQRA). Version 2.0. September, 2010 (Revised 2012). Prepared by: Contaminated Sites Division, Safe Environments Directorate. Cat.: H128-1/11-632E-PDF.

Health Canada. 2010b. Federal Contaminated Site Risk Assessment in Canada Part II: Health Canada Toxicological Reference Values (TRVs) and Chemical-Specific Factors. Version 2.0. September, 2010. Prepared by: Contaminated Sites Division, Safe Environments Directorate. Cat.: H128-1/11-638E-PDF.

Health Canada. 2010c. Federal Contaminated Site Risk Assessment in Canada. Part V. Guidance on Human Health Detailed Quantitative Risk Assessment for Chemicals ($DQRA_{CHEM}$). September, 2010. Prepared by: Contaminated Sites Division, Safe Environments Directorate. Cat.: H128-1/11-639E-PDF.

Health Canada. 2009. Federal Contaminated Site Risk Assessment in Canada. Part I. Guidance on Human Health Preliminary Quantitative Risk Assessment (PQRA). Version 2.0. May, 2009. Prepared by: Contaminated Sites Division, Safe Environments Programme.

Hrudey, S.E., Chen, W., C.G. Rousseaux. 1996. Exposure Routes and Bioavailability Factors for Selected Contaminants. I. Arsenic and III. Chromium and Chromium Compounds. In: S.E. Hrudey (Ed.), Bioavailability in Environmental Risk Assessment. CRC Press, Boca Raton, FL.

IARC. 2013. IARC Monographs on the Evaluation of Carcinogenic Risks to Humans. Volume 107. Polychlorinated and Polybrominated Biphenyls. Lyon, FR: International Agency for Research on Cancer. *In Press*.

Kelly, M.E., Brauning, S.E., Schoof, R.A., Ruby, M.V. 2002. Assessing Oral Bioavailability of Metals in Soil. Battelle Press, Columbus, Ohio.

Krewski, D., and Thomas, R.D. 1992. Carcinogenic mixtures. Risk Anal 12(1):105-113.

MCP. 1996. Massachusetts Contingency Plan: Risk Characterization and Evaluation, Massachusetts Department of Environmental Protection (MADEP), Bureau of Waste Site Cleanup, 1996/1997, Ch. 9. Cited in: ASTM, 2011.

Nagorsen, D.W. 1996. Opossums, Shrews and Moles of British Columbia. Volume 2: The Mammals of British Columbia. Royal British Columbia Museum Handbook. UBC Press. Cited in: Azimuth, 2012b.

NAS. 1983. Risk Assessment in the Federal Government: Managing the Process. National Academy of Science. National Academy Press, Washington, DC.

GNLDEC. Policy Directive. Management of Impacted Sites. PPD05-01. Government of Newfoundland and Labrador, Department of Environment and Conservation. February 22, 2005.

GNLDEC. 2005. Guidance Document for the Management of Impacted Sites. Version 1.01. Government of Newfoundland and Labrador, Department of Environment and Conservation. September, 2005.

OMOE. 2011. Rationale for the Development of Soil and Groundwater Standards for Use at Contaminated Sites in Ontario. April 2011. Prepared by: Standards Development Branch. Ontario Ministry of the Environment. PIBS 7386e01.

Oomen, A.G., Brandon, E.F.A., Swartjes, F.A. and Sips, A.J.A.M. 2006. How can information on oral bioavailability improve human health risk assessment for lead-contaminated soils? Implementation and scientific basis. RIVM report 711701042/2006.

Oomen, A.G., Hack, A., Minekus, M., Zeijdner E., Cornelis C., Schoeters G., Verstraete W., Van de Wiele T., Wragg J., Rompelberg C., Sips A., and Wijnen, J. 2002. Comparison of Five In Vitro Digestion Models To Study the Bioaccessibility of Soil Contaminants. Environmental Science & Technology. 36 (15): 3697 – 3705.

Palumbo-Roe, B., and Klinck, B. 2007. Bioaccessibility of arsenic in mine waste-contaminated soils: A case study from an abandoned arsenic mine in SW England (UK). J Environ Sci Health Part A 42:1251-1261.

Raymond, M., and Bergeron, J.M. 1986. Sexual dimorphism and morphometric variations in the ermine, Mustela erminea cicognanii. Can J Zool. 64:1966-1972. Cited in: Azimuth, 2012b.

Richardson, M. 1997. Compendium of Canadian Human Exposure Factors for Risk Assessment. 1155-2720 Queensview Dr., Ottawa, Ontario.

Ruby, M.V., Schoof R., Brattin, W, Goldade, M., Post, G., Harnois M., Mosby D.E., Castell S.W., Berti W., Carpenter M., Edwards D., Cragin C., and Chappell, W. 1999. Advances in Evaluating the Oral Bioavailability of Inorganics in Soil for Use in Human Health Risk Assessment. Environmental Science & Technology. 33 (21): 3697-3705.

Sample, B.E., Aplin, M.S., Efroymson, R.A., Suter, G.W., and Welsh, C.J.E. 1997. Methods and Tools for Estimation of the Exposure of Terrestrial Wildlife to Contaminants. Oak Ridge National Laboratory, Oak Ridge, TN. October, 1997.

Sample, B.E., D.M. Opresko, and G.W Suter II. 1996. Toxicological Benchmarks for Wildlife: 1996 Revision. Oak Ridge National Laboratory, Oak Ridge. ES/ER/TM-86/R3. http://www.esd.ornl.gov/programs/ecorisk/documents/tm86r3.pdf.

Sample, B.E., and Suter, G.W. 1994. Estimating Exposure of Terrestrial Wildlife to Contaminants. Oak Ridge National Laboratory, Environmental Sciences Division, Oak Ridge, TN.

Schoof, R.A. 2003. Guide for incorporating bioavailability adjustments into human health and ecological risk assessments at Department of Defense facilities. Part 2: Technical background document for assessing metals bioavailability. Update prepared for Tri-Service Ecological Risk Assessment Workgroup. June, 2003. http://web.ead.anl.gov/ecorisk/related/documents/Bioavail_Part_2-final.pdf.

SGE. 2003a. New Harbour Waste Disposal Site Environmental Testing. Final Report. Prepared for: Upper Trinity South Environment Committee and Department of Environment. Prepared by: SGE Acres Limited. February, 2003. P14418.00.

SGE. 2003b. New Harbour Part 1 Phase II Environmental Testing. Final Report. Prepared for: Upper Trinity South Environment Committee and Department of Environment. Prepared by: SGE Acres Limited. May, 2003. P14766.00.

SNC. 2012. Final Report. 2011 – 2012 Annual Report of Activities. Upper Trinity South (New Harbour) Waste Disposal Site. Submitted to: Department of Environment and Conservation. Submitted by: SNC Lavalin Inc. July, 2012.

Surfical Geology of Insular Newfoundland, Geological Survay Branch, Department of Mines and Energy, Government of Newfoundland and Labrador, Map 90-08, 1990.

Suter II, G.W. 2007. Ecological Risk Assessment. Second Edition. CRC Press.

Suter, G.W., Efroymson, R.A., Sample, B.E., and Jones, D.S. 2000. Ecological Risk Assessment for Contaminated Sites. Lewis Publishers, Boca Raton.

Suter, G.W. 1999. A framework for assessment of ecological risks from multiple activities. Human and Ecological Risk Assessment 5(2):397-413.

Suter, G.W. 1990. Endpoints for regional ecological risk assessments. Environ Man. 14(1):19-23. Cited In: CCME, 2006.

Suter, G.W. 1989. Ecological endpoints. Pages 2-1 in W. Warren-Hicks, B.R. Parkhurst, and S.S. Baker Jr. (eds.) Ecological Assessment of Hazardous Waste Sites: A Field and Laboratory Reference document. EPA 600/3-89/013. Corvallis Environmental Research Laboratory, Corvallis, OR. Cited In: Suter II, 2007.

Tryphonas, H., S. Hayward, L. O'Grady et al. 1989. Immunotoxicity studies of PCB (Aroclor 1254) in the adult rhesus (Macaca mulatta) monkey -- preliminary report. Int J Immunopharmacol. 11:199-206. Cited in: U.S. EPA, 1996.

Tryphonas, H., M.I. Luster, G. Schiffman et al. 1991a. Effect of chronic exposure of PCB (Aroclor 1254) on specific and nonspecific immune parameters in the rhesus (Macaca mulatta) monkey. Fund Appl Toxicol. 16(4):773-786. Cited in: U.S. EPA, 1996.

Tryphonas, H., M.I. Luster, K.L. White et al. 1991b. Effects of PCB (Aroclor 1254) on non-specific immune parameters in Rhesus (Macaca mulatta) monkeys. Int J Immunopharmacol. 13:639-648. Cited in: U.S. EPA, 1996.

U.S. EPA. 2007. Guidance for evaluating the oral bioavailability of metals in soils for use in human health risk assessments. United States Environmental Protection Agency. May, 2007. OSWER 9285.7-80.

U.S. EPA. 2005. Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities. Final. United States Environmental Protection Agency, Office of Solid Waste and Emergency Response (5305W). EPA530-R-05-006. September, 2005. http://www.epa.gov/osw/hazard/tsd/td/combust/riskvol.htm#volume1.

U.S. EPA. 2003. Attachment 1-3. Guidance for Developing Ecological Soil Screening Levels (EcoSSLs). Evaluation of Dermal Contact and Inhalation Exposure Pathways for the Purpose of Setting EcoSSLs. OSWER Directive 92857-55. November, 2003

U.S. EPA. 2000. Guidance for Assessing Chemical Contaminant Data for Use in Fish Advisories. Volume 2: Risk Assessment and Fish Consumption Limits. Third Edition. Office of Science and Technology, Office of Water, U.S. Environmental Protection Agency. EPA 823-B-00-008. November, 2000. http://water.epa.gov/scitech/swguidance/fishshellfish/techguidance/risk/upload/2009_04_23_fish_advice_volume2_v2cover.pdf.

U.S. EPA. 1999. Screening Level Ecological Risk Assessment Protocol for Hazardous Waste Combustion Facilities. Peer Review Draft. United States Environmental Protection Agency, Solid Waste and Emergency Response (5305W). EPA530-D-99-001A. August, 1999. http://www.epa.gov/osw/hazard/tsd/td/combust/ecorisk.htm.

U.S. EPA. 1998. Guidelines for Ecological Risk Assessment. EPA/630/R-95/002F. Risk Assessment Forum, U.S. Environmental Protection Agency, Washington, D.C.

U.S. EPA. 1997. Ecological Risk Assessment Guidance for Superfund: Process for Designing and Conducting Ecological Risk Assessments. Interim Final. United States Environmental Protection Agency, Solid Waste and Emergency Response. EPA 540-R-97-006. June, 1997. http://www.epa.gov/oswer/riskassessment/ecorisk/ecorisk.htm.

U.S. EPA. 1996. Aroclor 1254 (CASRN 11097-69-1). Integrated Risk Information System. United States Environmental Protection Agency. URL: http://www.epa.gov/iris/subst/0389.htm.

U.S. EPA. 1994. Ecological Risk Assessment Issue Papers. U.S. Environmental Protection Agency (U.S. EPA), Office of Research and Development. EPA/630/R-94/009.

U.S. EPA. 1993. Wildlife Exposure Factors Handbook. Volume I of II. Office of Health and Environmental Assessment, Office of Research and Development, U.S. Environmental Protection Agency. EPA/600/R-93/187.

WHO. 2003. Concise International Chemical Assessment Document 55. Polychlorinated Biphenyls: Human Health Aspects. http://www.inchem.org/documents/cicads/cicads/cicad55.htm#5.0.

WHO. 1992. Polychlorinated Biphenyls and Terphenyls (Second Edition). Environmental Health Criteria 140. International Programme on Chemical Safety, World Health Organization. http://www.inchem.org/documents/ehc/ehc/ehc140.htm.

Wragg, J., Cave, M., and Nathanail, P. 2007. A study of the relationship between arsenic bioaccessibility and its solid-phase distribution in soils from Wellingborough, UK. J Environ Sci Health Part A 42:1303-1315.