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**Human Health and Ecological Risk
Assessment and Ancillary Assessment
Activities: Lake Enon Former Mill Site, Enon,
Nova Scotia (FINAL)**

Lake Enon Former Mill Site, Enon, Nova Scotia
PID Nos. 15551369, 15340045, and 15340052



January 24, 2025 - 22-3723



January 24, 2025

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***Human Health and Ecological Risk Assessment and Ancillary Assessment Activities:
Lake Enon Former Mill Site, 2412 Loch Lomond Road
Parcel Identification Designation Numbers (PID Nos.): 15551369, 15340045, and
15340052 (Final Report)***

Dillon Consulting Limited (Dillon) is pleased to provide the final report for the Human Health and Ecological Risk Assessment (HHERA) and Ancillary Assessment Activities Report for the Lake Enon Former Mill property located at 2412 Loch Lomond Road in Enon, Nova Scotia, and identified by PID Nos. 155136, 15340045, and 15340052. The purpose of the additional assessment activities, as detailed herein, was reduce uncertainty and provide further understanding of the initial risk estimates calculated for contamination identified on the site or in Lake Enon, and to inform the development of the remedial approach and remedial design for the site.

Should you have any questions, please do not hesitate to contact us.

Sincerely,

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A handwritten signature in blue ink, appearing to read "N. Wambolt".

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Acronyms, Abbreviations, Definitions

| | |
|----------------------|--|
| ACCDC | Atlantic Canada Conservation Data Centre |
| AEC | Area of Environmental Concern |
| AEP | Alberta Environment and Parks |
| ALARA | As Low As Reasonably Achievable |
| ANZECC | Australian and New Zealand Environment and Conservation Council |
| AO | Aesthetic Objective |
| APEC | Area of Potential Environmental Concern |
| ARBCA | Atlantic Risk-Based Corrective Action |
| ARMCANZ | Agriculture and Resource Management Council of Australia and New Zealand |
| ASTM | American Society for Testing and Materials |
| ATSDR | Agency for Toxic Substances and Disease Registry |
| AUF | Area Use Factor |
| BC | British Columbia |
| BTEX | Benzene, Toluene, Ethylbenzene and Xylenes |
| BV | Bureau Veritas |
| BW | Body Weight |
| CCME | Canadian Council of Ministers of the Environment |
| COC | Contaminant of Concern |
| COPC | Contaminant of Potential Concern |
| COSEWIC | Committee on the Status of Endangered Wildlife in Canada |
| CSM | Conceptual Site Model |
| CSR | Contaminated Sites Regulations |
| CVOC | Chlorinated Volatile Organic Compounds |
| CWQG | Canadian Water Quality Guidelines |
| CWS | Canada Wide Standard |
| DNA | Deoxyribonucleic Acid |
| DO | Dissolved Oxygen |
| DOC | Dissolved Organic Carbon |
| DQRA _{CHEM} | Detailed Quantitative Risk Assessment for Chemicals |
| ECCC | Environment and Climate Change Canada |
| EcoSSL | Ecological Soil Screening Levels |
| EDI | Estimated Daily Intake |
| EHQ | Ecological Hazard Quotient |
| EPA | Environmental Protection Agency |
| EPC | Exposure Point Concentration |
| EQS | Environmental Quality Standards |
| ER | Exposure Ratio |
| ERA | Ecological Risk Assessment |
| ESA | Environmental Site Assessment |
| FCSAP | Federal Contaminated Sites Action Plan |

| | |
|---------|---|
| FDA | Food and Drug Administration |
| FEQG | Federal Environmental Quality Guideline |
| FIGQG | Federal Interim Groundwater Quality Guideline |
| GPS | Global Positioning System |
| HC | Health Canada |
| HHERA | Human Health and Ecological Risk Assessment |
| HHRA | Human Health Risk Assessment |
| HQ | Hazard Quotient |
| ICMM | International Council on Mining and Metals |
| ILCR | Incremental Lifetime Cancer Risk |
| ISQG | Interim Sediment Quality Guideline |
| IVBA | In Vitro Bioaccessibility Testing |
| LIDAR | Light Detection and Ranging |
| LMWPAH | Low Molecular Weight Polycyclic Aromatic Hydrocarbons |
| LOAEL | Lowest Observed Adverse Effect Level |
| LOE | Lines of Evidence |
| LOEC | Lowest Observed Effective Concentration |
| m bgs | Metres Below Ground Surface |
| MAC | Maximum Acceptable Concentration |
| Mbtoc | Metres Below Top of Casting |
| MDEP | Maine Department of Environmental Protection |
| MDEQ | Mississippi Department of Environmental Quality |
| MECCS | Ministry of Environment and Climate Change Strategy |
| MECP | Ministry of Environment, Conservation, and Parks |
| MW | Monitoring Well |
| NA | Not Applicable |
| NC | Not Calculated |
| NGA | No Guideline Applicable |
| NOAA | National Oceanic and Atmospheric Administration |
| NOAEL | No Observed Adverse Effect Level |
| NRC | National Research Council |
| NS | Nova Scotia |
| NS CSR | Nova Scotia Contaminated Sites Regulations |
| NS PSS | Nova Scotia Pathway Specific Standards |
| NSCS | Nova Scotia Contaminated Sites |
| NSDNRR | Nova Scotia Department of Natural Resources and Renewables |
| NSECC | Nova Scotia Department of Environment and Climate Change |
| NSLI | Nova Scotia Lands, Inc. |
| NTS | National Topographic System |
| ON | Ontario |
| ON MECP | Ontario Ministry of the Environment, Conservation, and Parks |
| PAH | Polycyclic Aromatic Hydrocarbons |
| PARCC | Precision, Accuracy, Representativeness, Comparability and Completeness |

| | |
|--------------------|--|
| PCB | Polychlorinated Biphenyls |
| PEL | Probable Effect Level |
| PHC | Petroleum Hydrocarbons |
| PID | Parcel Identification Designation Number |
| PIRI | Partnership for the Implementation of Risk-Based Corrective Action |
| PPE | Personal Protective Equipment |
| PQRA | Preliminary Quantitative Risk Assessment |
| PSS | Pathway Specific Standards |
| QA/QC | Quality Assurance/Quality Control |
| RAF | Relative Absorption Factor |
| RBA | Relative Bioavailability |
| RBCA | Risk-Based Corrective Action |
| RDL | Reportable Detection Limit |
| RPD | Relative Percent Difference |
| RMC | Royal Military College of Canada |
| ROC | Receptor of Concern |
| RQ | Risk Quotient |
| RSL | Regional Screening Level |
| SABCS | Science Advisory Board of Contaminated Sites in British Columbia |
| SAR | Species At Risk |
| SARA | Species At Risk Act |
| SF | Slope Factor |
| SoQG | Soil Quality Guideline |
| SoQG _E | Ecological Health-Based Soil Quality Guideline |
| SoQG _{HH} | Human Health-Based Soil Quality Guideline |
| SQG | Sediment Quality Guideline (freshwater aquatic life) |
| SSI | Supplemental Site Investigation |
| SSTL | Site-Specific Target Level |
| sVOC | Semi-Volatile Organic Compounds |
| SW | Surface Water |
| TCRL | Target Cancer Risk Level |
| TDI | Tolerable Daily Intake |
| THQ | Target Hazard Quotient |
| TOC | Total Organic Carbon |
| TPH | Total Petroleum Hydrocarbons |
| TRV | Toxicological Reference Values |
| TSS | Total Suspended Solids |
| TUF | Temporal Use Factor |
| UCLM | Upper Confidence Limit of the Mean |
| US EPA | United States Environmental Protection Agency |
| VOC | Volatile Organic Compounds |
| WHO | World Health Organization |
| WOE | Weight of Evidence |

Executive Summary

Dillon Consulting Limited (Dillon) was retained by Build Nova Scotia, on behalf of the Nova Scotia Department of Natural Resources and Renewables (NSDNRR), to conduct a Human Health and Ecological Risk Assessment (HHERA) for the Lake Enon Former Mill Property in Enon, Nova Scotia, herein referred to as the “site” or “subject property”.

The Lake Enon Former Mill (i.e., the site) is located in Enon, Nova Scotia along Loch Lomond Road, approximately 50 kilometers (km) southwest of Sydney in the Cape Breton Regional Municipality, and is situated adjacent to Lake Enon (**Figure 1, Appendix A**). The former mill site [Parcel Identification Designation Numbers (PID Nos.) 15551369, 15340045, 15340052] is owned by NSDNRR. The total property area of the site (i.e., all three PIDs) is approximately 141 hectares (ha). The area of the site impacted by historical site activities comprises a smaller, localized portion of the site consisting of twelve (12) Areas of Environmental Concern (AECs). Based on sampling within the property boundary completed to date, the contaminated area is generally limited to the portions of the site east and west of Loch Lomond Road, which intersects the site, and along the south shoreline of Lake Enon. The study boundary (or “study area”) for the HHERA encompasses approximately 51 ha of the total area of the former mill site. The HHERA land-based study boundary is shown in **Figure 2, Appendix A**. Potential off-site impacts to Lake Enon were also examined in the HHERA; however, the evaluation of potential impacts to other off site water bodies or land adjacent to the site, was outside the scope of this HHERA.

A draft HHERA was completed in October, 2023 which contained several recommendations to help address some uncertainties in the risk assessment. As part of an iterative risk assessment approach, a supplemental site investigation (SSI) was conducted between April and July 2024, and the data collected were used in the HHERA herein to reduce uncertainty and refine the initial risk assessment results. In addition, vegetation analytical data collected for the pre-design activities for on-site remediation (Dillon, 2024) were also considered in this final HHERA where relevant.

The HHERA considered potential risks to site visitors; vegetation and terrestrial invertebrates; herbivorous, insectivorous and carnivorous small mammals; herbivorous and insectivorous birds; freshwater aquatic life; amphibians; and the snapping turtle, via relevant exposure pathways to the identified contaminants of potential concern (COPCs). A summary of conclusions and recommendations for the HHERA are provided below. These conclusions and recommendations are based on the available data and assumptions used in the exposure and risk calculations. If site conditions and/or exposure assumptions change, the risk estimates should be re-evaluated to confirm the conclusions and recommended risk management measures for this assessment still apply.

The overall conclusions from the HHRA include:

- The THQ (target hazard quotient) for lead was exceeded for the toddler receptor (HQ=3.6). Risk estimates for lead for the child, teen and adult receptors were below the THQ based on the data and exposure assumptions applied.
- All the strontium HQs were below the THQ for all receptors.
- While there is a potential risk to toddlers in relation to lead, the HQ of 3.6 for the toddler is considered an overestimate. This is largely because it is extremely unlikely that toddlers would be present within the study boundary for any significant duration. The study boundary also has no unique features that would attract or encourage toddlers and their caregivers to be spending time in the lead-impacted portions of the study area. The HHRA conservatively assumed that a toddler would be on the site once per week for 6 months of the year. However, the likelihood of this occurring is considered negligible. The site and study boundary considered in the HHRA is not easy to access for the public and has a number of physical hazards such as unfenced waste rock piles, settling and tailings ponds. The study boundary also contains wetlands and dense forests in some areas. Much of the study boundary was difficult to traverse for Dillon field personnel and this would be even more the case for a toddler. It is highly unlikely that the study boundary would be used for recreational purposes, particularly with young children, when there are numerous other areas in the vicinity of the site and around the shores of Lake Enon that would be far more suitable for families engaging in recreational activities.

The overall conclusions from the ERA include:

- The potential for adverse effects to vegetation and soil invertebrate communities as a result of COPCs is within the study boundary is considered to be negligible to low within most areas of the site based on the available data. In areas where waste rock and tailings have been placed, the likelihood of potential risks is medium to high. The presence of waste rock and tailing and the lack of nutrients in the “soils” in these areas have most likely impacted vegetation in these areas and its ability to grow back/repopulate.
- There is the potential for individual level effects in the masked shrew, ruffed grouse, dark-eyed junco, based on the available data and assumptions used in the exposure and risk modelling. The likelihood of potential effects on local populations of small herbivorous mammals, carnivorous mammals, small herbivorous birds or insectivorous birds is considered to be negligible to low. Given the home and foraging range for the shrew would be encompassed by the areas having elevated metals and based on the elevated effects-based EHQs (ecological hazard quotients), there is a low to medium potential for effects on local populations of insectivorous small mammals with small home ranges (e.g., shrew). The remedial activities recommended for the HHRA (e.g., applying a cap to soils in excess of the human SSTL for lead) will reduce the potential risks to terrestrial ecological receptors.
- The potential for adverse effects in pelagic-feeding freshwater aquatic life communities as a result of contamination on the site is considered to be negligible based on the available data.

- For benthic species, exceedances over freshwater aquatic life sediment quality guidelines occurred in most sampling locations. While the contamination has been present for years and there may be some species acclimation/adaption, the degree to which this has occurred is unknown. The results of the benthic community, aquatic vegetation, and bivalve tissue analysis indicate that although COCs are present, the overall potential for adverse effects to benthic species as a result of site-related contamination is low.
- Based on the available site data and available amphibian toxicity data, the potential risk to amphibians is considered to be low to medium.
- While the snapping turtle could potentially be exposed to site-related contamination, their exposures would likely be limited due to the limited suitable habitat and food resources. In addition, the aquatic vegetation investigation revealed only marginal differences between COPC concentrations in near-field vegetation relative to far-field vegetation; therefore, potential uptake of site-related COCs from vegetation ingestion is expected to be limited. It is also unknown if the snapping turtle currently occurs within the study boundary as none were observed during field work.

Proposed risk management recommendations, including additional sampling needs, for the site and HHERA study boundary are as follows:

- The recommended soil SSTL (site-specific target level) for lead is 2100 mg/kg. This SSTL value is protective of all human receptors that were evaluated in the HHRA, including the toddler, under the exposure assumptions applied. The method or remedy by which the soil SSTL for lead will be achieved will be provided under separate cover from this HHERA report.
- Some miscellaneous drums and debris were noted by AEC 1 and south of AEC 12. Some of the drums were partially buried and appeared empty. While not a chemical exposure issue, these drums and debris could pose a physical risk to wildlife or someone walking in the area. It would be prudent to remove the visible debris and drums.
- Prevent the potential flow of water from the on-site culvert into Lake Enon.
- Prevent the movement of contaminated sediments from the tailings ponds/water courses into Lake Enon.
- Develop site-specific health and safety plan for site remediation workers, as required.
- If remedial activities result in freshly exposed waste rock / tailings, these areas should be evaluated for sulfide bearing materials for acid rock drainage potential as warranted. This can be done using a field screen.
- Given the wide-spread exceedances of lead in sediments in Lake Enon, conduct fish tissue sampling (filet and whole body) for metals to confirm fish tissue that may be ingested by humans and / or wildlife are not of concern. In addition, if hunting becomes a relevant exposure pathway in the future, it would be prudent to conduct game mammal and / or edible bird tissue sampling for species that are potentially harvested and consumed.

- If there are areas of fine tailings that are an open source of lead which could disperse and mobilize, resulting in increased concentrations within the study area boundary or off-site, these areas should be addressed in the remedial plan.
- Climate change impacts that could potentially increase concentrations of lead within the study boundary or off-site (e.g., decreased water in tailings ponds such that tailings become exposed, or increased flooding such that contamination spreads into surrounding areas or lakes) should be addressed in the remedial plan.

1.0

Introduction

Dillon Consulting Limited (Dillon) was retained by Build Nova Scotia, on behalf of the Nova Scotia Department of Natural Resources and Renewables (NSDNRR), to conduct a Human Health and Ecological Risk Assessment (HHERA) and Ancillary Assessment Activities for the Lake Enon Former Mill Property in Enon, Nova Scotia, herein referred to as the “site” or “subject property”. A draft HHERA was completed by Dillon (2023c) which contained several recommendations to help address some uncertainties in the risk assessment. As part of an iterative risk assessment approach, a supplemental site investigation (SSI) was conducted in April to July 2024, and the data collected were used in the HHERA herein to reduce uncertainty and refine the initial risk assessment results in the draft HHERA. In addition, vegetation analytical data collected for the pre-design activities for on-site remediation (Dillon, 2024) were also considered in this final HHERA where relevant.

The Lake Enon Former Mill is located in Enon, Nova Scotia along Loch Lomond Road, approximately 50 kilometers (km) southwest of Sydney in the Cape Breton Regional Municipality, and is situated adjacent to Lake Enon (**Figure 1, Appendix A**). The former mill site (Parcel Identification Designation Numbers [PID Nos.] 15551369, 15340045, and 15340052) is owned by NSDNRR. The total property area of the site (i.e., all three PID Nos.) is approximately 141 hectares (ha). The area of the site impacted by historical site activities comprises a smaller, localized portion of the site consisting of twelve (12) Areas of Environmental Concern (AECs). Based on sampling within the property boundary completed to date, the contaminated area is generally limited to the portions of the site east and west of Loch Lomond Road, which intersects the site, and along the south shoreline of Lake Enon. The study boundary (or “study area”) for the HHERA encompasses approximately 51 ha of the total area of the former mill site. The HHERA study boundary is shown in **Figure 2, Appendix A**. Photos of the site are provided in **Appendix B**. Potential off-site impacts to Lake Enon (AEC 8) were also examined as part of the HHERA. However, the evaluation of potential impacts to other off site water bodies or land adjacent to the site was outside the scope of this HHERA.

The findings presented in this report are based on information and analytical data obtained from the Phase I and II Environmental Site Assessments (ESAs), supplemental assessments (Dillon, 2022a; Dillon, 2022b; Dillon, 2023b), ancillary studies conducted in 2023, the pre-design activities report (Dillon, 2024) and the 2024 SSI completed as part of the current investigation, other reports and site records, information obtained during site visits, and discussions with the province and Build Nova Scotia.

1.1

Structure of the Report

The current section of this report (**Section 1.0**) details the objectives and scope of the HHERA. **Section 2.0** provides information related to the site background and characterization, including physical descriptions of the site, summaries of previous investigations, and site data available for the assessment. Site figures can be found in **Appendix A**, while site photos are provided in **Appendix B**. The

ancillary studies conducted to inform this HHERA are briefly described in **Section 3.0** and the full reports for each of these studies can be found in **Appendices C and D**. The Human Health Risk Assessment (HHRA) framework and outcomes are detailed in **Section 4.0**, and the Ecological Risk Assessment (ERA) framework and outcomes are detailed in **Section 5.0**. **Section 6.0** briefly describes climate change considerations relevant to the HHERA, and overall conclusions and recommendations from the HHERA are presented in **Section 7.0**. Risk management discussions relevant to the site are provided in **Section 8.0**. Limitations and closure conditions are described in **Section 9.0** and references are provided in **Section 10.0**. Additional information to support the HHERA (including analytical data tables, risk assessment modelling and the results of the 2024 SSI and associated studies) is provided in **Appendices E to L**.

1.2 Objectives and Scope

The overall study boundary for the HHERA is provided in **Figure 2, Appendix A**. The study boundary generally included the following APECs identified in previous reports (Dillon, 2022b) and shown on **Figure 2, Appendix A**:

- APEC 1: Tailings Pile/Pond;
- APEC 2: Waste Rock/Dump;
- APEC 3: Processing Area;
- APEC 4: Mill/Plant Area;
- APEC 5: Waste Rock/Dump;
- APEC 6: Tailings Pile/Pond;
- APEC 7: Settling Ponds;
- APEC 8: Lake Enon (Potential Receptor);
- APEC 9: Waste Rock/Dump;
- APEC 10: Tailings Disposal Area;
- APEC 11: Tailings Disposal Area; and
- APEC 12: Former Pad Area.

These above 12 APECs (Areas of Potential Environmental Concern) were identified as areas of the site anticipated to be impacted by historical site activities in the Phase I ESA. The results of the Phase II ESA and subsequent sampling, have confirmed contamination in these areas, and as such, they will be referred to as AECs from here on in. Photos and descriptions of the terrestrial habitat in each of the above AECs is provided in **Section 3.3.2 of Appendix D**.

This HHERA focused on analytical data for site soil, sediment, surface water, and groundwater collected within and adjacent to, the AECs. This included soil data collected within the study boundary and surface water and sediment data collected on site, and off-site within Lake Enon. Bivalve and aquatic vegetation tissue data from Lake Enon, and benthic community data from Lake Enon and three on-site watercourses, were also collected and investigated in the HHERA. Wetland vegetation analytical data

collected for the pre-design activities for on-site remediation (Dillon, 2024) were also considered in this final HHERA where relevant. Local background soil data (from within the subject property), and background sediment and surface water data (from Lake Enon) were also collected outside of the expected areas of contamination. Vegetation data collected for the Sample locations are provided in **Figure 3, Appendix A**.

The scope of the study was to evaluate on-site contamination in addition to potential impacts to Lake Enon. The evaluation of near-shore sediments in Lake Enon, and the evaluation of other off-site water bodies, or any other land based properties adjacent to the site was outside the scope of the HHERA as per direction from Build NS.

The HHERA for the site was conducted in accordance with the Nova Scotia Contaminated Sites Regulations (NS CSRs) and associated Ministerial Protocols, where applicable. Provincial guidelines were applied where possible (i.e., the Nova Scotia Pathway Specific Standards [NS PSS] and the Atlantic Risk-Based Corrective Action [ARBCA] guidelines). In addition, Federal Contaminated Sites Action Plan (FCSAP), Health Canada (HC), Environment and Climate Change Canada (ECCC) guidance materials and applicable provincial and federal protocols and guidelines were also considered, where relevant (see **Section 4.2.4** and **5.2.8** for details of the guidelines applied in the human and ecological risk assessments, respectively).

The scope of the project conducted and reported on herein included conducting a Human Health and Ecological Risk Assessment in addition to several ancillary studies listed below.

- Supplemental Sampling Program (2023);
- Benthic Habitat Assessment;
- Aquatic Habitat and Wetland Assessment;
- Terrestrial Habitat and Species at Risk (SAR) Assessment;
- Supplemental Site Investigation (2024); and
- Benthic community study.

Details pertaining to the 2023 Supplemental Sampling Program can be found in **Appendix C**. A Health and Safety plan was completed prior to any field work. Details of the Benthic Habitat Assessment, Aquatic Habitat and Wetland Assessment, and Terrestrial Habitat and SAR Assessment can be found in **Appendix D**. Details of the 2024 SSI (including bivalve and aquatic vegetation tissue analytical data), and the benthic community study can be found in **Appendix L**.

The objectives and scope of this Human Health and Ecological Risk Assessment are as follows:

- Update the initial draft HHERA report completed by Dillon in October 2023, based on the 2024 supplemental site investigations and studies conducted on the site to date. This included

- screening relevant new data collected to identify any changes to the contaminants of concern (COC)¹ for the HHERA;
- Where possible and if necessary, estimate exposures (as part of the exposure assessment step) using standard receptor parameters and appropriate statistical values to provide a realistic assessment of the potential risks;
 - As part of the toxicity assessment (also called the hazard or effects assessment), identify exposure limits or toxicological reference values (TRVs) (i.e., concentration of a chemical not anticipated to be associated with adverse health effects (or associated with a low level of effects in the ERA)) for each COC; and
 - Calculate, as part of the risk characterization step, the potential for adverse health risks associated with the estimated exposures and toxicity for each COC for each receptor and provide risk management recommendations, as needed, based on the results of the HHERA.

The site is zoned for commercial land use, and neighbouring properties are zoned for residential and commercial land use. No known potable wells are currently located on the site, nor are they reasonable anticipated to be constructed in the future. However, the surrounding properties are serviced by individual potable wells with nearest residential building located approximately 100 metres (m) north/cross gradient to the site boundary. Given neighbouring properties are zoned for residential land use, potable wells could potentially be constructed closer to the site in the future.

Two grain size samples were collected from different areas of the site during the Phase II ESA. The results indicated one sample was classified as fine-grained and one sample was classified as coarse-grained. As such, the Nova Scotia Tier II Pathway Specific Standards (PSS) for both fine- and coarse-grained soils have been considered. Therefore, for the purpose of the HHERA, the site was considered to be commercial and potable, with both fine and coarse-grained soil conditions.

The HHERA evaluates current and reasonably foreseeable future exposures associated with the site, assuming the anticipated future land use, existing engineering controls and site conditions remain consistent with current land use, as is anticipated.

¹ Health Canada (2021a) uses the term COPC (contaminant of potential concern) to identify the final list of contaminants to be assessed in the human health risk assessment. FCSAP (2012a) uses the term COPC to identify the initial list of substances considered, and COC (contaminant of concern) to identify the final list of chemicals to be assessed in the ecological risk assessment at the end of the problem formulation stage. For this HHERA, the term COC is used to identify the final list of chemicals retained in the HHERA at the end of the problem formulation stage.

Background and Site Characterization

The Lake Enon Former Mill (i.e., the site) is located in Enon, Nova Scotia along Loch Lomond Road, approximately 50 km southwest of Sydney in the Cape Breton Regional Municipality and is situated adjacent to Lake Enon. The former mill site (PID Nos. 15551369, 15340045, 15340052) is owned by NSDNRR. As reported in the Phase II ESA (Dillon, 2022b), significant deposits of celestite, the principal source of strontium, were discovered from the Lower Windsor rock group in Enon in the early 1960s. The site was developed in the mid-1960s and was operational from 1969 through 1975 or 1976 by Kaiser Celestite Mining (Kaiser). During this time, ore was reportedly sourced from an on-site pit, as well as a quarry to the north of the site. Concentrated ore was then sent to Kaiser's chemical plant in Point Edward, NS to be converted to different strontium compounds. Site photos can be found in **Appendix B** and in **Section 3.2** and **Section 3.3** of **Appendix D**.

In 1977, the site was purchased by Yava Mines Ltd. (Yava). Between 1979 and 1981, Yava operated the milling site to process lead ore from a nearby mine. The site was then acquired by Novex Mining and Exploration (Novex). From 1983 to 1984, Novex operated the mill site to process barite ore. In 1988, Lodestone operated the mill site to process a magnetite bulk sample from Bass River. Buildings and processing equipment associated with the former mill operations were reportedly removed from the site in the mid-1990s.

The total property area of the site is approximately 141 ha (**Figure 1, Appendix A**). The area of the site anticipated to be impacted by historical site activities comprises a smaller, localized portion of the site consisting of the AECs as identified in **Section 1.2**. The contaminated area is limited to the northwest portion of the property along Lake Enon. The study boundary for the HHERA has an area of approximately 51 ha (**Figure 2, Appendix A**). Land areas within the property boundaries but outside of the study boundary have been considered as background locations for soil. Potential off-site impacts to Lake Enon (AEC 8) were also examined as part of the HHERA.

The study boundary contains notable features including two settling ponds (AEC 1 and AEC 6), two infilled settling ponds (AEC 10 and AEC 11), a tailings pond (AEC 6), two waste rock piles from previous mining activities, and a concrete pad from previous mining activities with the Loch Lomond Road running through the study area. Some miscellaneous drums and debris near AEC 1 and southwest of AEC 12 were noted by Dillon field crews (see **Photo 21** and **Photo 22, Appendix B**). There is a steep slope along portions of the tailings pond which would potentially limit access by some wildlife and humans. The settling ponds and tailings pond were artificially constructed for the mine and were not originally aquatic habitat. Currently these areas provide some habitat for small fish. During the habitat study, hundreds of minnows were observed in the tailings pond (AEC 6) and small fish (approximately 3 inches long) and smaller minnows in the settling ponds in AEC 7.

Four watercourses and three wetlands were identified during the site visit by Dillon field staff in July 2023. Two of the watercourses were confirmed to have fish, one was considered to provide seasonally accessible fish habitat and the other was considered unlikely to provide fish habitat. In general, site drainage is towards Lake Enon. Additional information can be found in the aquatic habitat assessment report (**Appendix D, Section 3.2**).

Lake Enon is identified as a habitat of potential ecological concern since it is within 200 m of the site (i.e., it borders the site immediately north) (Atlantic PIRI, 2012) and a drainpipe connects the settling ponds to the lake. In addition to the settling ponds, the site contains large wet areas with widespread cattail growth and water-logged soil.

The study boundary is well vegetated with trees and shrubs to the west of AECs 1, 2 and 11 and also to the east of Loch Lomond Road (with the exception of the waste rock/dump [AEC 5]); however, the central areas of the study boundary are less vegetated. Much of the areas within/in the vicinity of AECs 1, 2, 10 and 11 are wetlands and the western areas of the study boundary are densely forested which would limit the use of these areas of the site by humans and some wildlife (see **Figure 4, Appendix D**). Information on potential terrestrial habitat is provided in **Section 3.3.2, Appendix D**.

2.1 Physical Setting

2.1.1 Water Supply/Groundwater Usage

During the Phase II ESA field program from May 17 to 19, 2022 (Dillon, 2022b), groundwater was encountered at depths ranging from 1.16 m below top of casing (mbtoc) (monitoring well [MW] 3) to 5.78 mbtoc (MW4). Based on the measured groundwater elevations, the generalized groundwater flow direction in the study area is interpreted to be convergent towards Lake Enon. There are currently no potable groundwater wells on site; however, the surrounding properties are serviced by individual potable wells with nearest residential building located approximately 100 m north/cross gradient to the site.

2.1.2 Regional Geology and Topography

To describe the regional physiography and expected hydrogeological conditions beneath the subject property, the following documents were reviewed as part of the Phase I ESA (Dillon, 2022a):

- Stea, R.R., Conley, H., and Brown, Y. (compilers) 1992: Surficial Geology of the Province of Nova Scotia; Nova Scotia Department of Natural Resources, Map 92-3, Scale 1:500,000;
- Keppie, J.D. (compiler) 2000: Geological Map of the Province of Nova Scotia; Nova Scotia Department of Natural Resources, Minerals and Energy Branch, Map ME 2000-1, Scale 1:500,000; and

- S.M. Barr, C.E. White (compilers) 2017: Bedrock Geology Map of the Grand Narrows Area, NTS 11F/15, Cape Breton, Inverness, Richmond and Victoria Counties, Nova Scotia; Nova Scotia Department of Natural Resources, Map ME 2017-014, Scale 1:50,000.

The surficial geology of the subject area consists of stony till plain and drumlins. Till is stony, sandy and consists of material derived from local bedrock sources, ranging in thickness from 2 m to 20 m. Drumlin facies are siltier due to erosion and incorporation of older till units by glaciers. The regional bedrock geology of the site is mapped as variable. The Uist formation is present at the northwest portion of the site, northwest of the former pond. Between the former pond and tailings pond, the Loch Lomond formation is present. To the southeast of Lake Enon, the Enon formation is present, followed by the Grantmire formation, and then by the Chisholm Brook Plutonic Suite - monzogranite. A small area of the Chisholm Plutonic Suite - granodiorite is present with the Loch Lomond formation on the southwest portion of the site. Bedrock mapping of the area also identified occurrence of strontium on the subject property, south of Lake Enon and north of the former tailings disposal pond.

The topography of the site is flat to rolling with many surface boulders; drumlins-elongate or oval hills veneered by stony till with underlying multiple till layers. Regional topography suggests the regional shallow groundwater flow is likely west to north/northwest towards Lake Enon. Historical reports (see **Section 2.3**) noted that a portion of the site (PID No. 15340052) was leveled and graded between 1995 and 1997. It is noted that some of the historical lagoons and pits have been infilled and the characteristics of the fill material are not known. The local shallow groundwater flow direction may vary from the regional context and be influenced by backfilled areas with coarse-grained materials, which may provide a more permeable conduit for groundwater flow when compared to the lower permeability of the native soils.

2.1.3 Stratigraphy

Due to the nature of the site, there is significant variability in soil stratigraphy depending on the area. Additionally, the grade of the site is even more variable depending on the area within the study boundary. During the Phase II ESA it was noted that soil stratigraphy in the area generally consisted of a thin layer of organics underlain with silt and gravel with varying amounts of sand and clay, with cobbles atop bedrock. Bedrock was encountered in the background area at a depth of 3.05 meters below ground surface (m bgs).

2.1.4 Climate

Current climate conditions at the site can be characterized by examining the historical data. Climate data were obtained from ClimateAtlas.ca using the historical period 1976-2005. All data are for the Cape Breton Regional Municipality which includes the Lake Enon site (ClimateAtlas.ca, 2023):

- Average annual temperature: 6.1 °C;
- Coldest month: February: -5.5 °C;
- Hottest month: September: 14.2 °C;

- Annual average precipitation: 1394 mm;
- Number of frost days (days with coldest temperature below 0°C): 150.5 days;
- Number of summer days (days above 25°C): 13.7 days; and
- Dry days (number of days without rain or snow): 186.1 days.

The site is situated within the maritime boreal climate zone and as shown above, is characterized by variable weather, warm summers, cold winters, and moderate precipitation.

2.2 Land Use Considerations

The site is zoned for commercial land use, and neighbouring properties are zoned for residential and commercial land use. The site is currently vacant with no buildings present. A concrete pad is located on the eastern side of Loch Lomond Road. To date, metal impacts in soil are delineated laterally on the southwest corner of the site study boundary and on the east portion of the study boundary. Lateral delineation of metals impacts in soil has not been achieved to the north or south and would require off-site sampling.

No potable wells are currently located on site; however, the surrounding properties are serviced by individual potable wells with the nearest residential building located approximately 100 m north/cross gradient to the site.

2.3 Previous Investigations

2.3.1 Historical Reports (Pre-2022)

As noted in the 2022 Phase I ESA (Dillon, 2022a), the following reports were available for review:

- A.D. Hudgins, Milado Mines (1957) Limited Mine Drilling Report, Loch Lomond Area – Cape Breton, NS;
- Lura Corporation Limited, Milado Mines (1957) Limited Report. (1963);
- J.H. Fowler (1991) Barite, Celestite and Fluorite in Nova Scotia;
- Jacques, Whitford, and Associates Limited. (1983) Yava Mines Review, Cape Breton, NS;
- Dillon. (2018) Flooding Assessment Task 1 - Initial Assessment - Loch Lomond Road, Cape Breton; and
- Nova Scotia Transportation and Infrastructure Renewal (2018) Lake Enon Drainage Improvement Plan.

As noted in the 2022 Phase I ESA (Dillon, 2022a), not all reports contained information relevant to the ESA; therefore, the following summary of relevant information was provided:

- Float (i.e., loose pieces of rock that are not connected to an outcrop) containing mineralizations of galena, sphalerite, barite, and celestite were identified on the subject property. Diamond drilling (for Milado Mines, 1957) was completed on-site in 1963 to test the results of geological

- mapping, and previously completed geophysical surveys and geochemical testing. Three boreholes were reportedly completed on-site. Findings indicated that it was not considered feasible to undertake additional diamond drilling in the Loch Lomond Area until favorable targets were further delineated by gravity surveys;
- Kaiser operated the site from approximately 1969 to 1975 for the purposes of mining and milling celestite (a mineral consisting of strontium sulfate). The site contained several open pits, one of which was eventually used for the disposal of tailings, and a facility for milling the ore. Maps provided in this review display infrastructure including a pump, crusher building, transfer toner, concentrator, sub-station, fuel oil tank (22,700 L capacity), thickener tank, water storage tank, sewage treatment plant, and pump house. The facility also contained a laboratory and several sheds. Infrastructure was located west of Loch Lomond Road (formerly Enon Road). The former tailings pond and former settling pond are located west of this infrastructure. A process water reclaim pond was shown located between the tailings disposal pond and the two settling ponds. A seeded dump area to the east of Loch Lomond Road, opposite the former mining pit was displayed;
 - In 1979, Yava reactivated the mine until 1981, with the purpose of milling lead ore from a nearby mining site (Salmon River), located approximately 13 km northeast of the subject property. The tailings disposal pond used by Kaiser was utilized by Yava and was altered to accommodate a higher storage capacity. In 1981, Yava was placed in receivership resulting in withdrawal from the site. Yava reportedly did not conduct any stabilization or restoration of the tailings;
 - Disposal of tailings was conducted between 1981 and 1983 by use of a discharge line to the east of the tailings pond. Drainage was facilitated to the northwest corner of the subject property, where slime accumulation had occurred. Soil samples were collected to complete a sieve analysis, and slime samples were collected for geotechnical characteristics analysis (i.e., water content, liquid limit, plastic limit, unconfined pressure, and remoulded pressure). The majority of the tailings surface was categorized as sand and silty sand. The northwest corner of the tailings area contained slime deposits from internal drainage. The thickness of tailings from the disposal area was determined to range from 1.52 m to 2.44 m. The water table in the tailings area was encountered at depths ranging from 0.61 m to 0.76 m;
 - In 1983 until 1984, Novex Mining Exploration acquired the subject property for the purposes of milling and processing barite ore;
 - In 1988, the mill facility was used by Lodestone to process magnetite bulk samples from Bass River.
 - Prior to 1995, Industrial Estates Limited (IEL), a provincial government economic development company, assumed ownership of the mill site. This did not include the tailings disposal pond or part of the open flooded pit;
 - Between 1995 and 1997, Kelly Rock Limited removed all on-site buildings and leveled and graded the site;
 - A 2018 flooding assessment was completed by Dillon, investigating backups of water onto Loch Lomond Road during significant rainfall events. Since the closure of the site, a portion of

- underground piping previously used to manage stormwater on-site during mining and process operations had collapsed, contributing to this backup. Surface water samples were collected in the former tailings disposal pond, the tertiary pond, near the shore of Lake Enon, and near a road culvert on Loch Lomond Road. Samples were analyzed for aluminium, ammonia, cadmium, iron, and lead and results were compared to Canadian Water Quality Guidelines (CWQG) and Nova Scotia Contaminated Sites (NSCS) Table 3 Guidelines. Concentrations of lead in both samples collected from the tailings disposal pond exceeded the CWQG and NSCS guidelines concentrations of analyzed parameters in samples collected in the tertiary pond and near the shore of Lake Enon were below CWQG and NSCS. Exceedances of both criteria in all analyzed parameters, with the exception of ammonia, were observed in the surface water sample collected near the road culvert, which were attributed to significant corrosion of the culvert; and
- A Phase I ESA, completed by ADI Nolan Davis in 1995, is referenced in the 2018 flooding assessment noted above; however, this report was not able to be provided to Dillon for review. The flooding assessment report (Dillon, 2018) noted that the 1995 Phase I ESA had identified the potential for polychlorinated biphenyls (PCB) related issues and buried process reagents. Though no further details were provided, it has been assumed that these issues would be in the areas of former pits/lagoons at the site.

Based on the above, the following potential or actual environmental concerns were identified:

- Mining operations with regards to celestite;
- Milling operations and associated reagent use (floatation, coagulants, flocculants, etc.) with regards to processing celestite (selenium), lead, and barium, as well as select organic compounds;
- Potential acid rock drainage;
- Potential PCB related issues; and
- Potential buried process reagents.

2.3.2

Phase I ESA (Dillon, 2022a)

Dillon was commissioned by Build Nova Scotia (formerly Nova Scotia Lands Inc. (NSLI)) to complete a Phase I ESA of the property with the objective of assessing whether the site is or may be subject to actual or potential contamination. The report identified twelve APECs as well as contaminants of potential concern (COPC) for each of those APECs in various media (soil, groundwater, sediment, and surface water). **Table 2-1** is adapted from the Phase I ESA (Dillon, 2022a) and offers some additional insight into the conditions surrounding each APEC. Notable observations surrounding site conditions are wet areas of soil near APECs 10 and 11, large piles of waste rock in APECs 5 and 2, and the presence of artificially constructed settling ponds (which were not originally aquatic habitat).

Table 2-1: Summary of APECs and Site Conditions. Adapted from Phase I ESA (Dillon, 2022a).

| APEC No. | APEC | Comments |
|-----------------|------------------------------------|--|
| 1 | Tailings Pile/Pond | Waste storage, aerial mapping shows surficial water present in this area |
| 2 | Waste Rock/Dump | Waste/tailings soils area |
| 3 | Processing Area | Processing of materials, heavy equipment storage, petroleum storage |
| 4 | Mill/Plant Area | Processing of materials, heavy equipment storage, petroleum storage |
| 5 | Waste Rock/Dump | Waste/tailings soils area |
| 6 | Tailings Pile/Pond | Waste storage, aerial mapping shows surficial water present in this area |
| 7 | Settling Ponds | Waste storage, aerial mapping shows surficial water present in this area |
| 8 | Lake Enon (Potential Receptor) | Lake Enon is identified as a potential receptor. Given the nature of the site conditions, sediment and surface water quality should be evaluated |
| 9 | Waste Rock/Dump | Waste/tailings soils area |
| 10 | Tailings Disposal Area (filled in) | Waste/tailings storage. Historical plans indicate a pond was previously located here |
| 11 | Tailings Disposal Area (filled in) | Waste/tailings soils area |
| 12 | Former Pad Area | Former pad area used for storage, waste rock pile area |

2.3.3 Phase II ESA (Dillon, 2022b)

Dillon was commissioned by Build Nova Scotia (formerly NSLI) to complete a Phase II ESA of the Lake Enon Former Mill property. The Phase II ESA included the advancement of eight boreholes, installation of seven monitoring wells, completion of twelve test pits, advancement of several shallow test holes, and soil, groundwater, surface water and sediment sampling. The report identified exceedances in soil and various other media, and a risk assessment was recommended.

2.3.4 Supplemental ESA (Dillon, 2023b)

Following the Phase II ESA, Dillon was contracted by BNS to complete a supplemental ESA at the site. This supplemental ESA included: preparation of detailed site contour maps using LIDAR to support future work, installation of a new well (MW5-D), and additional groundwater, surface water, and background soil sampling. The report concluded with recommendations for a hydrological and sediment transport study, the completion of a benthic habitat and benthic community assessment for the lake and the completion of a risk assessment.

2.3.5 HHERA Problem Formulation (Dillon, 2023a)

Following the Phase II ESA and collection of additional samples required for the risk assessment, a HHERA problem formulation was completed for metals in site soils only (i.e., no other environmental media or potential contaminants). The problem formulation identified lead and strontium as COCs for human receptors and cadmium, copper, lead, strontium, and zinc as COCs for ecological receptors. Further sampling of environmental media (i.e., soil, sediment, surface water, and groundwater) was recommended for delineation purposes and to incorporate into a full HHERA. Additionally, a

hydrological and sediment transport study was recommended to determine if contaminated sediments and fines are being transported off-site to the adjacent lake. An aquatic habitat assessment for Lake Enon (including benthos, fish, and amphibians) was also recommended along with a formal Species at Risk (SAR) assessment to reduce uncertainties related to SAR use of the site. The Dillon (2023a) problem formulation was updated in the initial HHERA to incorporate the results of the 2023 field studies and to include the new data collected as part of the 2023 supplemental sampling program.

2.4 Data Availability and QA/QC

2.4.1 Background Data

The background soil data selected for HHERA screening purposes were compiled from the sampling program in the Phase II ESA (Dillon, 2022b), the supplemental ESA for the property (Dillon, 2023b), and the additional site investigation work conducted in 2023 (**Appendix C**). Soil metals data were collected within the Lake Enon property boundary, but outside the areas considered to be potentially contaminated. These samples were considered to represent local background conditions. In total, twenty-five (25) background samples collected from within the property but outside the areas considered potentially impacted were analyzed for metals. Local background samples were collected at various depths and distances from the site AECs, ranging from approximately 100 m to 800 m from the nearest AEC (**Figure 3, Appendix A**). Local background soil data can be found in **Appendix E**.

The areas selected to represent local background were identified by reviewing soil analytical results at all depths outside of the AECs, in addition to reviewing site photos, information obtained during site visits, site elevations (see **Figure 9, Appendix A**), and considering the likelihood of windblown dust to impact the areas outside of the AECs. Soil samples collected outside of the study boundary at different depths were reviewed to identify whether there were large differences in surface and sub-surface soil concentrations (e.g., by comparing analytical results for samples at 0 to 0.15 m and 0.15 to 1.0 m from the same sampling location).

Background lead soil concentrations ranged from 7 to 170 mg/kg, while background strontium concentrations ranged from <5 to 47 mg/kg. Background sample location TP35, which is located approximately 200 m from identified site AECs, had a lead soil concentration of 63 mg/kg at both the surface (0 to 0.15 m) and subsurface (0.15 to 1 m). The similar lead concentrations at the different depths provides support to the assumption that this area would be unlikely to be associated with wind-blown dusts and can be considered representative of background concentrations. While the maximum local background lead soil concentration of 170 mg/kg (TH53, collected approximately 300 m from the study boundary at a depth of 0 to 0.15 m; no analytical results from deeper depths) seems high compared to some of the other background data (particularly samples to the west of the study boundary), the general area is known to be enriched with lead. The closest samples to TH53 had lead concentrations of 110 mg/kg (TH52), 74 mg/kg (SSS41), and 98 mg/kg (SS42). Sample TH75 had a lead concentration of 140 mg/kg at 0 to 0.5 m and a concentration of 24 mg/kg at 0.3 to 0.75 m. Lead

analytical results for samples in the vicinity of TH75 include: 53 mg/kg at TH47; 18 mg/kg at TH51; 63 mg/kg at SSS38 and 7.1 mg/kg at SSS43.

The Mineral Occurrence Database indicated that the celestite ore mined in the Lake Enon Pit contained average lead concentrations of approximately 5,000 mg/kg. However, these samples would have been collected at much deeper depths than are relevant for human and ecological exposures. In addition, the ore sample results reported in this database would have undergone a strong acid digest that would provide higher lead concentrations than the more typical “available metals” acid digestion that is used in standard soil analytical methods. While these ore sample results are not appropriate for use as background concentrations in the HHERA, they indicate that lead can be naturally elevated in some areas of the overall site.

In addition to local background soil data, one background surface water and one background sediment sample were collected from Lake Enon and analyzed for metals. Background analytical data for metals in groundwater that were collected in 2022 were also available from one monitoring well (MW1) located to the east of the study boundary.

Three (3) background (i.e., far-field) samples were collected in Lake Enon for each of the following: benthic invertebrate community, aquatic vegetation tissue, and bivalve tissue.

Background soil, groundwater, sediment, and surface water analytical data considered in the HHERA can be found in **Appendix E**. Background (i.e., far-field) benthic community, aquatic vegetation, and bivalve tissue data can be found in **Appendix L**.

Laboratory certificates of analysis were reviewed for internal quality assurance/quality control (QA/QC) metrics (i.e., method blanks, certified reference materials, spiked blanks, and matrix spikes).

Overall, no significant data quality issues were identified that would affect the use of the site data in the HHERA. Laboratory certificates of analysis for data collected as part of the current supplemental sampling program can be found in **Appendix C**, while the laboratory certificates of analysis for previously collected data can be found in **Appendix F**.

2.4.2 Site Data

The site data selected for the HHERA were compiled from the Phase II ESA field program (Dillon, 2022b), supplemental ESA (Dillon, 2023b), the additional site investigation work conducted in 2023 (**Appendix C**), and data from the 2024 SSI and the pre-design activities report (**See Section 3.5**). Site soil, groundwater, sediment, and surface water analytical data considered in the HHERA can be found in **Appendix E** (all data inclusive up to 2024). Benthic community, aquatic vegetation, and bivalve tissue data from the 2024 SSI is provided in **Appendix L**.

Soil

One hundred and thirty-three (133) soil samples (including duplicates) collected from various depths within the study boundary were analyzed for metals. Sixteen (16) soil samples were analyzed for benzene, toluene, ethylbenzene, and xylenes (BTEX) and petroleum hydrocarbons (PHCs), with not all parameters being analyzed in all samples. Twenty-three (23) soil samples were analysed for polycyclic aromatic hydrocarbons (PAHs). Thirteen (13) soil samples were analysed for volatile organic compounds (VOCs) and semi-volatile organic compounds (sVOCs), and six (6) soil samples being analyzed for PCBs.

All of the soil data collected from within the study boundary at relevant depths were included in the HHERA, with the exception of the soil data collected at depth from within the study boundary in late September 2023 as part of the supplemental ESA. The September 2023 soil samples were collected to help delineate soil contamination within the study boundary and to provide support for the areas selected to represent local background for the study. While the surface soil data collected in September 2023 from within the study boundary and in background areas were included in the relevant HHERA screening, the soil data collected from deeper depths were not, given the late timing of the receipt of sample analytical results relative to the project timeline and given these data would not impact the assessment of the receptors selected for quantitative evaluation in the HHERA.

Groundwater

Eleven (11) groundwater samples were collected from seven (7) on-site monitoring wells during 2022. Each sample was analyzed for metals, with a sub-set of the samples being analyzed for BTEX, PAHs, VOCs and sVOCs.

Surface Water

A total of twenty-two (22) surface water samples (including a duplicate sample) were collected from Lake Enon and the settling/tailings ponds within the study boundary. Each sample was analyzed for general chemistry and metals, and a sub-set of samples were analyzed for BTEX and PHCs, PAHs, VOCs and sVOCs. In addition, nine (9) surface water samples were collected from wetland areas within the study boundary and analyzed for metals and four (4) surface water samples (including one duplicate) were collected from on-site watercourses and analyzed for metals.

Sediment

A total of forty-three (43) sediment samples (including duplicates) were collected from Lake Enon and the settling/tailings ponds within the study boundary. Each of the samples were analyzed for metals, and a sub-set of samples were analyzed for BTEX and PHCs, PAHs, VOCs and sVOCs. In addition, ten (10) sediment samples were collected from wetland areas within the study boundary and analyzed for metals and three (3) samples were collected from on-site watercourses and analyzed for metals.

Benthic Invertebrate Community

A total of nine (9) benthic invertebrate community samples were collected from Lake Enon (near-field) and the watercourses within the study boundary and evaluated for various benthic invertebrate community indices as discussed in **Appendix L**.

Aquatic Vegetation Tissue

A total of nine (9) aquatic vegetation samples were collected from Lake Enon close to the mine site (i.e., near-field) and analyzed for metals and moisture as discussed in **Appendix L**.

Bivalve Tissue

A total of nine (9) bivalve tissue samples were collected from Lake Enon close to the mine site (i.e., near-field) and analyzed for metals and moisture as discussed in **Appendix L**.

2.4.3**Data Quality Assurance and Control**

A program to ensure data quality assurance and control (QA/QC) was implemented throughout the Phase II ESA (Dillon, 2022b), the supplemental ESA (Dillon, 2023b) and the additional site investigation work conducted in 2023 (**Appendix C**) and 2024 (**Appendix L**). The QA/QC program consisted of the following elements:

- Collection of samples using protocols consistent with Dillon Standard Environmental Field Procedures and/or industry standards;
- Use of dedicated sampling equipment and/or adherence to established equipment cleaning protocols, where applicable;
- Use of laboratory supplied containers;
- Collection of blind field duplicates; and
- Implementation of laboratory QA/QC procedures including analysis of reference standards, laboratory blanks and replicates.

Validation criteria were established that required the analytical data to have an acceptable and documented level of precision, accuracy, representativeness, comparability, and completeness (the PARCC criteria). The precision of the data for the samples collected was evaluated by calculating the Relative Percent Difference (RPD) between the original samples and its duplicate when the samples had concentrations greater than five (5) times the laboratory Reportable Detection Limit (RDL).

Laboratory certificates of analysis for data collected as part of the 2023 supplemental sampling program can be found in **Appendix C**, and the 2024 SSI can be found in **Appendix L**, while the laboratory certificates of analysis for previously collected data can be found in **Appendix F**.

2.4.4

Bioaccessibility Data

To aid in calculating site-specific oral RBA (relative bioavailability), five soil samples were submitted for *in vitro* bioaccessibility (IVBA) testing for lead. The IVBA data were used in the exposure assessment step of the HHRA to aid in estimating the oral bioavailability of lead in soils. Additional details are provided in **Section 4.4.2**. Results of the IVBA testing can be found in **Appendix G**.

3.0

Ancillary Studies

Several ancillary studies were completed to further characterize contamination on site and inform the HHERA conducted herein. A brief summary of these studies is presented below. Additional details of the Supplemental Sampling Program conducted in 2023 can be found in **Appendix C**, while details of the benthic habitat assessment, aquatic habitat and wetland assessment, and terrestrial habitat and species at risk assessment are provided in **Appendix D**. Details from the 2024 SSI, and data relevant for the HHERA from the pre-design activities report (Dillon, 2024) are provided in **Appendix C** and **L**.

3.1

Supplemental Sampling Program (2023)

Dillon undertook a supplemental sampling program in 2023 which included additional soil, surface water, and sediment sampling within the property, as well as off-site surface water and sediment sampling in Lake Enon. The purpose of the additional sampling was to further characterize the site, inform the HHERA, and inform the development of the remedial approach for the site. The sampling program is briefly described below, and further details pertaining to the background, methodology, and results can be found in **Appendix C**. Sample numbers collected as part of the supplemental sampling program have been included in the total sample numbers previously provided in **Sections 2.4.1 and 2.4.2**.

3.1.1

Soil Sampling

Shallow surface soil sampling was completed by Dillon at thirty-four (34) on-site locations (sample IDs TH44 through TH78; **Figure 3, Appendix A**) on June 20 and 23, 2023, and on September 20, 2023. Shallow surface soil samples were collected using hand tools with samples collected at select intervals between 0 and 1.5 m bgs. On September 20, 2023, a supplemental test pit program was conducted to further assess soil conditions on the subject property. Dillon retained B. Curry & Sons Construction Ltd. from Sydney, NS to conduct the test pitting. A total of seven (7) test pits (TP34 through TP40, including background TP35) were advanced to depths ranging from 2.45 to 4.0 m bgs using an excavator. Select samples were submitted to BV Labs in Sydney, NS for analysis of metals (including mercury), with the remaining soil samples held for potential future analysis. Due to observations of debris (machine parts) and visual/olfactory evidence of petroleum hydrocarbon impacts observed at one test pit location (TP34), two select soil samples (TP34 0.15-1.0 m bgs and TP34 2.0-2.45 m bgs) were submitted for analysis of BTEX and TPH.

Analytical results from the supplemental soil sampling program are presented in **Appendix C** and discussed in the HHERA (**Section 4.0**) and the ERA (**Section 5.0**). Laboratory certificates of analysis for data collected as part of the supplemental sampling program are presented in **Appendix C**.

3.1.2 Sediment and Surface Water Sampling

On July 4, 2023, surface water and sediment sampling were completed at fifteen (15) locations (SW11 through SW25 and SED1 through SED25, respectively) in Lake Enon. Dillon commissioned the services of a boat with Commercial Diving & Marine Services Atlantic to assist with collection of the surface water and sediment samples. A description of the sediment, including texture, consistency, colour, odour, presence of biota, and appearance was recorded (through field notes, photographs, and video) at each sampling location. Field water quality measurements were also recorded at each surface water sampling location. Samples were shipped to BV Labs in Sydney, NS for analysis of metals (including mercury), and/or total organic carbon and grain size.

On July 12, 2023, Dillon collected a surface water sample from a pipe discharging to Lake Enon (SW-D) during a precipitation event (i.e., the sample was collected during heavy rain and after approximately 17.6 millimeters (mm) of rain had fallen). Total suspended solids were not detected in the sample collected at SW-D during the rainfall event. Based on the two (2) samples collected from this discharge point, the rate at which sediments/fines may be flowing from the site into Lake Enon does not appear to be significantly increased during a heavy rainfall event.

Four additional surface water and sediment samples (WETLAND #1-A, WETLAND #1-B, WETLAND #2 and WETLAND #3) were collected from on-site wetlands on July 26, 2023. Samples were submitted to BV Labs in Sydney, NS for total suspended solids and/or metals analysis.

Grain size analysis, undertaken at twelve (12) sediment sampling locations, indicated most locations consisted of fine-grained sediment (i.e., 8 out of 12). Most of the locations that consisted of coarse-grained sediment (i.e., SED12, SED14, SED23 and SED25) were located near the shoreline.

Analytical results from the supplemental sediment and surface water sampling program are discussed in the HHRA (**Section 4.0**) and the ERA (**Section 5.0**). Laboratory certificates are presented in **Appendix C**.

3.2 Benthic Habitat Assessment

As part of the benthic habitat assessment, a Dillon biologist reviewed the diver video and photographs that were taken during the sediment sampling program to determine substrate conditions and available habitat for relevant receptors within the near shore areas of Lake Enon.

In general, the sediment of Lake Enon within the area that was assessed is soft and appeared to be mostly comprised of silt and sand. Rocky substrate with a coating of silt was observed near the SED 13 sampling location (located close to the shore, adjacent to the site). The vegetation was typical of freshwater lakes in Nova Scotia and was dominated by common pond weeds and aquatic grasses. Floater mussels (*Pyganodon* sp.) were observed at many of the sediment sampling sites, both embedded in the substrate and observed as shell debris.

Further details pertaining to the background, methodology, and results of the benthic habitat assessment, in addition to representative photos from each sampling location can be found in **Appendix D, Section 3.1**.

3.3 Aquatic Habitat and Wetland Assessment

An aquatic habitat and wetland delineation assessment was conducted to inform the HHERA. The purpose of the assessment was to identify watercourses and wetlands within the study boundary. In addition, the assessment was conducted to verify the presence of suitable habitat for amphibians and/or aquatic receptors and to map the boundaries of potential wetlands so that adverse impacts can be alleviated during remediation. The assessment is briefly described below, and further details pertaining to the background, methodology, and results can be found in **Section 3.2, Appendix D**. The locations of the watercourses and wetlands identified within the study boundary can be found in **Figure 4, Appendix D**.

3.3.1 Watercourse and Fish Habitat Field Survey

A fish habitat suitability survey was conducted for watercourses within the study boundary to assess whether they contained suitable fish habitat features. An aquatic habitat assessment form was used for the survey, which includes the collection of data for the characterization of the watercourses (e.g., physical habitat characteristics to meet the standards of potential applications required for NSNRR/Fisheries and Ocean Canada) and *in-situ* water quality readings (e.g., temperature, pH, conductivity, and dissolved oxygen).

During the aquatic habitat surveys, staff in the field also recorded the locations of culverts and water flow directions using a handheld GPS to support a high-level understanding of site flow paths and characteristics.

Four watercourses and three wetlands were identified within the study boundary and assessed during the site visit on July 25 and 26, 2023, which were conducted within two and three days of significant precipitation. A drainage ditch was also observed.

Results from the survey indicated that two of the four watercourses (WC1 and WC2) were confirmed to be suitable for fish habitat (fish were observed in these watercourses). Watercourse 3 (WC3) may provide seasonably accessible fish habitat, while watercourse 4 (WC4) is unlikely to provide suitable fish habitat since this watercourse is blocked downstream. WC2 and WC3 are likely to support fish passage and/or foraging between Lake Enon and the human-made settling ponds.

An engineered drainage ditch between the site access road and the settling pond at AEC 6 did not contain water during the field assessment. Given the significant rainfall prior to the site visit and the

high-water stage observed at other locations, it is considered unlikely that the drainage ditch provides fish habitat and/or fish passage to adjacent watercourses and ponds.

A summary of the watercourse assessment parameters collected in the field are provided below in **Table 3-1**. Photos of the watercourses and additional information on the watercourse study are provided in **Section 3.2.1, Appendix D**.

Table 3-1: Watercourse Assessment Data Summary

| Parameter | WC1 | WC2 | WC3 | WC4 |
|------------------------------|----------------------------------|----------------------------------|--|---|
| Wetted Width (m) | 3.0 | 2.0 | 0.50 | 2.0 |
| Channel Width (m) | 4.0 | 3.0 | 1.0 | 3.0 |
| Depth (m) | 0.30 | 0.25 | 0.20 | 0.20 |
| Substrate | Small gravel | Boulder | Boulder | Small gravel |
| Water Temperature (°C) | 17.5 | 24.5 | 25.6 | 17.7 |
| Dissolved Oxygen (DO) (mg/L) | 9.82 | 7.59 | 6.87 | 8.39 |
| Dissolved Oxygen (DO) (%) | 102.6 | 91.1 | 84.1 | 88.1 |
| Conductivity (µS/cm) | 109 | 216 | 214 | 77 |
| pH | 6.85 | 7.18 | 7.19 | 7.02 |
| Suitability for Fish Habitat | Confirmed (i.e., fish observed). | Confirmed (i.e., fish observed). | May provide seasonally accessible fish habitat | Unlikely to provide suitable fish habitat –blocked downstream |
| Assessment Date | 2023-07-26 | 2023-07-26 | 2023-07-26 | 2023-07-26 |

3.3.2 Wetland Field Delineation

A Dillon professional trained in wetland identification and delineation performed a reconnaissance-level survey of the study boundary for unmapped wetlands. Three wetlands were assessed and delineated within the study boundary during the site visit conducted between July 25 and 26, 2023. In addition, a fourth wetland (WL4) was assessed but not field delineated.

A summary of the observations at the wetlands are provided below in **Table 3-2**. Photos of the wetlands and additional information, including dominant vegetation identified at each wetland, are provided in **Section 3.2.2, Appendix D**.

Table 3-2: Summary of Site Wetlands

| Wetland ID | Wetland Type | Approximate Size (ha) | Landscape Position | Landform | Water Flow Path |
|-------------------|--------------|-----------------------|--------------------|----------|-------------------------|
| Wetland 1 (WL1) | Swamp/ Marsh | 1.33 | Lentic | Fringe | Bidirectional-non-tidal |
| Wetland 2 (WL2) | Swamp/ Marsh | 0.21 | Terrene | Basin | Paludified |
| Wetland 3 (WL3) ‡ | Swamp/ Marsh | ~3.8 | Terrene | Basin | Outflow |
| Wetland 4 (WL4)* | Swamp | ~2.9 | Terrene | Fringe | Outflow |

* Not field-delineated.

‡ Partially field delineated.

3.4 Terrestrial Habitat and Species at Risk Assessment

3.4.1 Terrestrial Habitat Assessment

A terrestrial habitat general assessment was conducted within the study boundary between July 25 and 26, 2023 by Dillon field staff. Photos within and adjacent to each of the AECs are provided in **Section 3.3.2, Appendix D**.

The site is within the Bras d'Or Lowlands Ecodistrict of the Northumberland/Bras d'Or Ecoregion of Nova Scotia (Neily et al., 2017). In general, the ecoregion is characterized by its proximity to the more temperate waters of the Northumberland Strait and the Bras d'Or Lakes, compared to either the Atlantic or Bay of Fundy (Neily et al., 2017). Locally, the site is within the Spruce Pine Hummocks ecoelement (Neily et al., 2017).

The terrestrial landscape has been influenced by the anthropogenic historical use of the site. Many of the forested areas on and adjacent to the site have previously been harvested and are dominated by early successional and tolerant tree species and are in different stages of regeneration. Most of the forested areas are dominated by conifer trees (mainly Balsam fir (*Abies balsamia*), white pine (*Pinus strobus*), black spruce (*Picea mariana*)); however, some areas of mixed wood forests contain maple and birch trees along with conifer trees as dominant tree species. Some areas previously used as settling ponds have developed into swamp and marsh wetland.

Incidental wildlife observed during the site visits conducted in July and September of 2023 included: the Pickerel Frog (*Lithobates palustris*), Wood Frog (*Lithobates sylvaticus*), White-tailed Deer (*Odocoileus virginianus*), Northern Leopard Frog (*Lithobates pipiens*) and snails (*Gastropod asp.*).

3.4.2 Species at Risk Assessment

To identify a preliminary list of SAR and species of conservation concern (SoCC) that may use the site, relevant federal and provincial guidance was reviewed along with applicable regulations and legislation. The following definitions of SAR and SoCC applied to the assessment:

- Species at Risk (SAR): A species that is determined to be endangered, threatened, or vulnerable/special concern by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC), Nova Scotia Endangered Species Act (NSESAs), or the federal Species at Risk Act (SARA); and
- Species of Conservation Concern (SoCC): those species that are not SAR but are identified as regionally vulnerable or imperilled by the Atlantic Canada Conservation Data Centre (AC CDC) (i.e., those species with AC CDC S-ranks of S1: critically imperilled in the province; S2: imperilled in the province; and S3: vulnerable in the province).

Data was gathered from AC CDC, previous reports, and the species at risk registry, with special attention to applicable recovery strategies, action plans, and management plans. The AC CDC report included 104 historical records of rare and endangered flora and fauna comprised of eighteen (18) species within a 5 km radius of the site centre, and of these records there were ten (10) species considered to be SAR, as defined above, for this assessment.

Following the desktop exercise, a Dillon biologist conducted a reconnaissance-level site visit to document the general terrestrial habitat present within the site. Data obtained from the site visit was used to assess the site's suitability for SAR identified in the desktop assessment (see **Table 3-3** below) and was used as a line of evidence in the ERA (**Section 5.2.4**).

Table 3-3: Suitable Habitat Descriptions for Potential SAR

| Species | Suitable Habitat Description | Available Suitable Habitat within the Site | Species on site? |
|--|---|---|----------------------------|
| Barn Swallow <i>Hirundo rustica</i> | Typically inhabit open areas near human settlements and land uses including parks, ball fields, golf courses and agricultural fields where they forage for flying insects. These birds will typically construct their nests on human-made structures, and rarely in more natural locations such as cliffs, caves, or hollowed trees (COSEWIC, 2021). Feeds primarily on flying insects but may occasionally take insects from the ground or on vegetation (NatureServe Explorer, 2023). | Suitable nesting habitat is not present within the site. Suitable habitat is adjacent to the site and the species could potentially forage on the site. | AC CDC records within 5 km |

| Species | Suitable Habitat Description | Available Suitable Habitat within the Site | Species on site? |
|--|--|---|-----------------------------------|
| Chimney Swift <i>Chaetura pelagica</i> | Aerial foragers and tend to concentrate near water where flying insects are abundant (ECCC, 2022a). | Suitable nesting habitat is not present within the site. Suitable habitat is adjacent to the site and the species could potentially forage on the site. | AC CDC records within 5 km |
| Olive-sided Flycatcher <i>Contopus cooperi</i> | Nests in open, forested areas, often with many conspicuous perches (i.e., tall trees or snags alongside open areas) (ECCC, 2016). Diet is comprised almost entirely of flying insects (NatureServe Explorer, 2023). | Potential suitable habitat in forested areas (AEC 1, 4, 10 and 12) | AC CDC records within 5 km |
| Rusty Blackbird <i>Euphagus carolinus</i> | Breeds in the boreal forest, in habitat characterized by coniferous-dominated forests adjacent to wetlands (EC, 2015). Rusty Blackbird diet includes invertebrates, seeds, and fruits (NatureServe Explorer, 2023). | Potential to be present in coniferous forests, unlikely to be present in the AECs. | AC CDC records within 5 km |
| Little Brown Myotis <i>Myotis lucifugus</i> | <p>Hibernation: Subterranean features, such as caves, abandoned mines, hand-dug wells, cellars, tunnels, rock crevices or tree root hollows where light and noise levels are low (ECCC, 2018). Hibernacula typically contain sections that have relatively stable temperatures (2-10°C) and stable, high humidity levels (>80 %) (ECCC, 2018).</p> <p>Roosting: Treed and forested habitat and man-made structures. Bats preferentially roost in older forest stands with higher snag densities compared to young forests (ECCC 2018).</p> <p>Diet: Include mainly flying insects, while the Northern Myotis may also glean prey from plants or the forest floor (NatureServe Explorer, 2023).</p> | Suitable roosting habitat is not present on the assessed areas of the site; bats may forage within the site on insects. | AC CDC records within 5 km |
| Northern Myotis <i>Myotis septentrionalis</i> | | | |
| Tricolored Bat <i>Perimyotis subflavus</i> | | | |
| Canada Lynx <i>Lynx canadensis</i> | Lynx forage in habitats that are suitable for hare (habitat with hardwood browse and softwood cover); and maternal dens are typically situated in habitat that includes coarse woody debris (NSLRT, 2006). Feeds primarily on small birds and mammals, particularly the snowshoe hare. Home range for male Canada Lynx averages 15 to 30 km ² but may be much larger when prey is scarce. Mean population density ranges from 2 to 9 per 100 km ² (NatureServe Explorer, 2023). | The site is not located within the Cape Breton Island Lynx Range. Potential to be present in coniferous forests, unlikely to be present in the AECs. | AC CDC records within 5 km |

| Species | Suitable Habitat Description | Available Suitable Habitat within the Site | Species on site? |
|---|---|--|----------------------------|
| Snapping turtle <i>Chelydra serpentina</i> | Slow-moving water with a soft mud bottom and dense aquatic vegetation (ECCC, 2020a). Overwintering sites require water that is shallow enough for the turtle to reach the surface to breathe, but deep enough that it will not freeze to the bottom and mud deep enough for the turtle to bury itself (ECCC, 2020a). Eggs are generally laid on sand or gravel banks near the water, in locations where vegetation is absent or sparse (ECCC, 2020a). Diet includes a variety of vertebrates, invertebrates, and plants (NatureServe Explorer, 2023). | Potential suitable habitat within settling ponds and Lake Enon (AECs 6, 7 and 8) | AC CDC records within 5 km |
| Atlantic Salmon - Eastern Cape Breton population <i>Salmo salar</i> pop. 4 | Requires rivers or streams that are generally clear, cool and well-oxygenated (COSEWIC, 2010). | Suitable habitat not present within the site. | AC CDC records within 5 km |
| New Jersey Rush <i>Juncus caesariensis</i> | Bogs and fens in a geographically restricted area of southeastern Cape Breton Island, Nova Scotia (ECCC, 2022b). | Suitable habitat not present within the site. | AC CDC records within 5 km |
| Blue Felt Lichen <i>Pectenia plumbea</i> | Hardwood trees with mature bark and high humidity in woodlands (ECCC, 2022c). | Suitable habitat not present within the site. | AC CDC records within 5 km |
| Boreal Felt Lichen - Atlantic pop. <i>Erioderma pedicellatum</i> | Cool, humid, forests containing Balsam Fir (<i>Abies balsamea</i>) (ECCC, 2020b). | Suitable habitat not present within the site. | AC CDC records within 5 km |

Bold denotes SAR with potential suitable habitat on the site.

Shaded cells denote SAR with potential suitable habitat near the site and that may incidentally be present on the site.

Overall, the assessment indicated the potential presence of two SAR on the site, as follows:

- Olive-sided flycatcher (*Contopis cooperi*); and
- Snapping turtle (*Chelydra serpentina*).

Further details pertaining to the background, methodology, and results of the terrestrial habitat and species at risk assessment can be found in **Section 3.3, Appendix D**.

3.5 Supplemental Site Investigation (2024)

Dillon conducted an SSI in 2024 which included additional surface water and sediment sampling, benthic invertebrate community sampling, and aquatic vegetation and bivalve tissue sampling. The purpose of the SSI was to reduce uncertainties identified in the initial HHERA and refine the risk assessment results.

A full summary of the 2024 SSI is provided in **Appendix L** and the 2024 data are summarized in **Appendix E**. The results of the 2024 SSI are incorporated into the HHERA herein where applicable and sample numbers collected as part of the 2024 SSI have been included in the total sample numbers previously provided in **Sections 2.4.1 and 2.4.2**.

3.6 Pre-Design Activities for On-Site Remediation (Dillon, 2024)

Vegetation analytical data from Wetland #3 collected for the pre-design activities were available and are also considered in this final HHERA, where relevant. These data are summarized and discussed in **Appendix L**.

The pre-design activities report also contained a Hydrological and Sediment Transport Study. The primary purpose of the study was to assess if contaminated soil/sediments and/or fines are being transported off site to Lake Enon via total suspended solids (TSS) within surface run-off. To assess a range of drainage conditions on-site, a typical rain event (1:2-year) and an extreme rain event (1:100-year) were evaluated based on at least 10 years of precipitation data. A drought year was also examined for comparison purposes. Using the results of the hydrological and sediment transport modeling, a discussion of potential sediment diffusion and dispersion were discussed.

Results of the Hydrological and Sediment and Transport study suggested that in a typical year, TSS contributions to contaminated sediment being transported off-site to Lake Enon would be considered de minimis. However, in a 1:100-year scenario, the model simulated suspended solids entering Lake Enon over a range of particle sizes. In instances where contaminated sediments are transported into Lake Enon, it is possible, given their particle size distribution, that they could be transported approximately across half of Lake Enon during the 1:2-year event and across the entire length of the lake during the 1:100-year event. While in a typical year, the TSS contributions to contaminated sediment being transported off-site into Lake Enon would be considered de minimis, there is the potential in extreme weather situations, for sediments to transport further across the lake. As such, for the benthic community, benthic invertebrate tissue and aquatic vegetation tissue discussions, sample locations were referred to as near-field and far-field rather than site and background, as it could not be distinguished whether sediments in the far-field have been potentially impacted from historical site contamination.

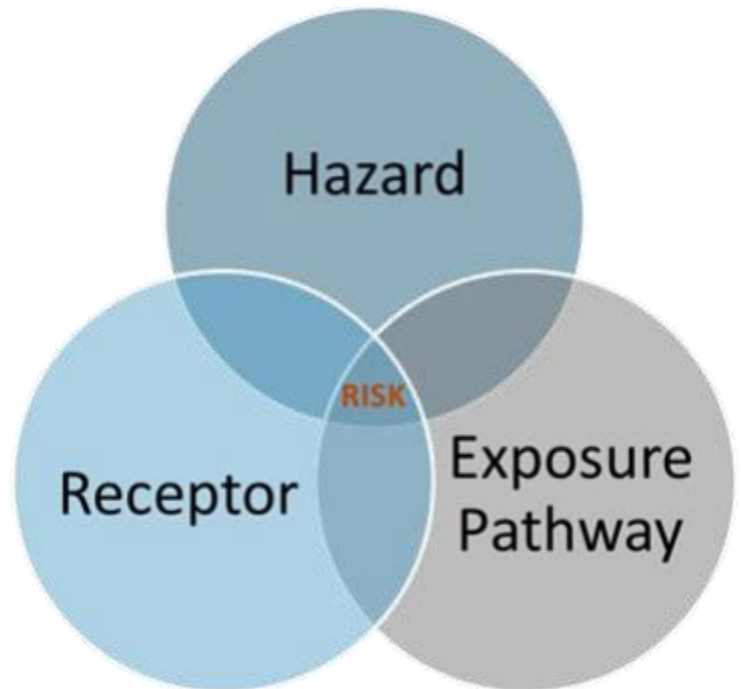
4.0

Human Health Risk Assessment

4.1 Human Health Risk Assessment Framework, Purpose, and Scope

Risk assessment is a systematic quantitative or qualitative process for interpreting the potential adverse effects associated with an activity, situation, or an exposure. The framework and methods used for the HHRA for the Lake Enon Former Mill site are described in this section and its associated subsections.

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). In order for a potential risk to exist, three things must be present: a contaminant of potential concern (or hazard), a receptor, and an exposure pathway that links the chemical constituent to the receptor, as presented in the graphic to the right. If one of these elements is missing, potential risk does not exist. If all three elements are present, then the degree of risk may be estimated and compared to target risk levels to identify whether or not unacceptable levels of risk are present and remedial action or risk management is required.



The HHRA for the Lake Enon Former Mill site was conducted in accordance with the following Canadian regulatory guidance documents, where applicable:

- Federal Contaminated Site Risk Assessment in Canada. *Guidance on Human Health Preliminary Quantitative Risk Assessment (PQRA)*. Version 3.0. March 2021. Health Canada Contaminated Sites Division, Safe Environments Directorate (Health Canada, 2021a);
- Federal Contaminated Site Risk Assessment in Canada. *Toxicological Reference Values (TRVs)*. Version 3.0. March 2021. Health Canada Contaminated Sites Division, Safe Environments Directorate (Health Canada, 2021b);
- Federal Contaminated Site Risk Assessment in Canada. *Overview of Health Canada Guidance Documents Related to Human Health Risk Assessment of Federal Contaminated Sites*. March

2021. Health Canada Contaminated Sites Division, Safe Environments Directorate (Health Canada, 2021c);

- Federal Contaminated Site Risk Assessment in Canada. *Part V. Guidance on Human Health Detailed Quantitative Risk Assessment for Chemicals (DQRA_{CHEM})*. September 2012. Prepared by: Health Canada Contaminated Sites Division, Safe Environments Directorate (Health Canada, 2010);
- *Canada Wide Standard for Petroleum Hydrocarbons in Soil: Scientific Rationale*. Supporting Technical Document. PN 1399. ISBN 978-1-896997-77-3 PDF (Canadian Council of Ministers of the Environment [CCME], 2008); and
- *CCME Protocol for the Derivation of Environmental and Human Health Soil Quality Guidelines* (CCME, 2006).

The HHRA also utilized relevant guidelines and approaches provided in the Nova Scotia Contaminated Sites ministerial protocols (Atlantic PIRI, 2022; NSECC, 2022). NSECC has adopted environmental quality benchmarks derived by the CCME, and other regulatory authorities in Canadian or international jurisdictions. Where appropriate or necessary, the Human Health Risk Assessment (HHRA) also considered human health risk assessment guidance and procedures developed and endorsed by other regulatory agencies such as the US EPA (Environmental Protection Agency). In this HHRA, the US EPA Regional Screening Levels (RSL) (US EPA, 2023) were used as a guideline source for COPC screening, consistent with Health Canada (2021a) PQRA guidance, in the absence of federal or provincial benchmarks.

An HHRA typically consists of five main steps or phases, as outlined below and in **Figure 4-1**.

Problem Formulation: The data gathered and evaluated in this step provides information regarding the history and layout of the subject site, possible exposure scenarios and pathways, identification of receptors of potential concern, identification of contaminants of concern, other specific issues of concern to be addressed, and the development of a conceptual site model.

Exposure Assessment: The exposure assessment of an HHRA involves estimation of exposure of hypothetical human receptors to each of the COPCs.

Toxicity (Hazard) Assessment: This step determines an exposure limit or toxicological reference value (i.e., concentration of a chemical not expected to be associated with adverse health effects or a concentration associated with an acceptable risk level) for each of the COPCs.

Risk Characterization: Integrates the exposure and toxicity (hazard) assessments to estimate the likelihood of potential 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 toxicity (hazard) assessment.

Risk Management Recommendations: determination of the need for corrective action, and development of site-specific risk management objectives and/or criteria (such as site-specific target levels, or SSTLs), if required.

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 **Sections 4.2** through **4.5**.

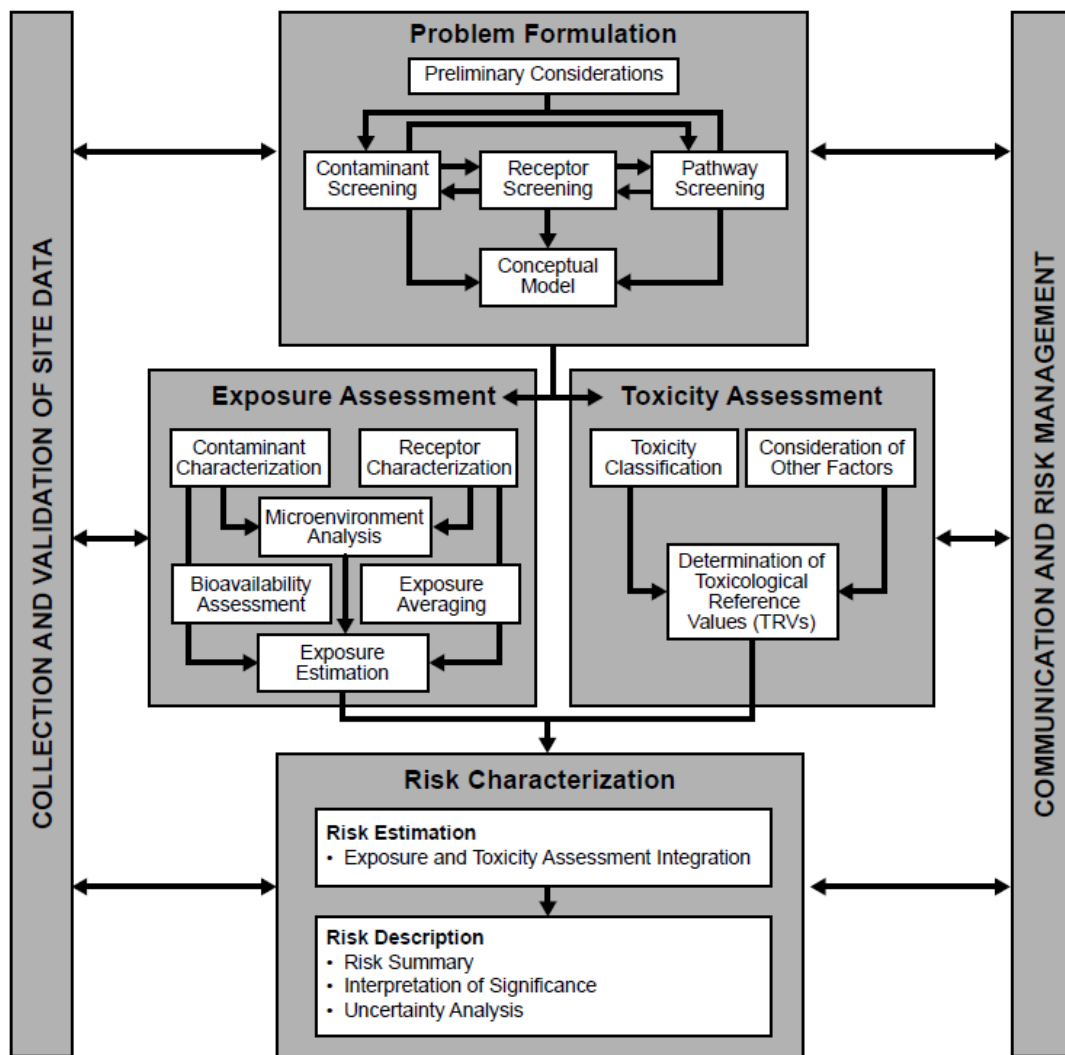


Figure 4-1: Human Health Risk Assessment Framework (from Health Canada, 2010)

4.2 Problem Formulation

The problem formulation step of an HHRA is an information gathering step that helps to plan and focus the risk assessment. The data gathered and evaluated in this step provides information regarding the history and layout of the subject site, possible exposure scenarios and pathways, identification of receptors of potential concern, identification of contaminants of concern, and other specific issues of concern to be addressed.

Key tasks requiring evaluation within the problem formulation step include the following:

- Site characterization;
- Selection of receptors;
- Selection of exposure pathways and scenarios;
- Identification of COPCs; and
- Development of a conceptual site model (CSM).

An initial human health problem formulation was completed for the HHRA by Dillon (2023a) in May 2023, which recommended additional sampling and the inclusion of additional site media. The problem formulation herein incorporates the other site media, and the 2023 supplemental sampling and ancillary studies conducted on the site to confirm and/or update the selected receptors, pathways, and contaminants of concern (COC) for the HHRA.

Outcomes of the problem formulation are provided in the following sections.

4.2.1 Site Characterization

The site characterization step provides a description of the site including information regarding the history and layout of the site and site characteristics that are pertinent to understanding and assessing potential exposures on the site.

Site characterization for the study boundary was previously provided in **Section 2.0**. Site photos are provided in **Appendix B** and within **Section 3.3.2 of Appendix D**.

4.2.2 Selection and Characterization of Receptors

A human receptor is a hypothetical person (e.g., infant, toddler, child, adolescent, or adult) who resides, visits, or works in the area being investigated and is, or could potentially be, exposed to the chemicals identified as being of potential concern. General physical/physiological and behavioural characteristics specific to the receptor type (e.g., body weight, breathing rate, amount of soil, and food consumed) 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. While HHRA guidance in some jurisdictions continues to differentiate receptors based on gender, Health Canada HHRA guidance has moved away from distinguishing between genders for the selection of human receptors.

Since people have varying physical/physiological 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 COPCs 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 that are considered to be either the most sensitive, or receive the greatest exposures from the 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.

In an HHRA, it is commonplace to select toddler receptors for the evaluation of chemicals that are not considered to be carcinogens (i.e., threshold response chemicals) or that have TRVs available that are based on a chemical's non-carcinogenic effects. This is because toddlers typically receive greater chemical exposures, via all pathways and routes, on a relative body weight basis, when compared to other human receptor classes.

For chemicals that are considered to be carcinogens, or that have TRVs available that are based on a chemical's carcinogenic effects, it is necessary that HHRAs evaluate such substances over an appropriate period of time, as the development of cancer is a long-term process that may take many years to manifest (i.e., latency). For this reason, a special type of hypothetical receptor called a "lifetime" or "composite" receptor is evaluated when potential carcinogenic risks require consideration in a HHRA.

Based on the current and reasonably anticipated future land use identified for the site, the following potential human receptors (i.e., people who may be exposed to COPCs from the site) were selected:

- **Site Visitor (toddler, child, adolescent and adult age classes)** – the most likely human receptor for the site are site visitors (all ages except infants) using the land for recreational purposes (walking, hiking, etc.). Infants are excluded from assessment as it is extremely unlikely infants would be present with their caregivers on the site. Furthermore, HHRAs do not typically assess infant receptors in relation to contaminated sites, unless the sites are residential or high use recreational.

There are currently no buildings on site and construction work is not expected to occur on-site in the future. However, should intrusive/remediation work be conducted on the site, workers could potentially come in contact with COPCs in soils at all depths. The evaluation of potential exposures and risks to workers who may participate in site remediation work that involves soil excavation, grading, stockpiling, and other soil handling activities, at some point in the future, is beyond the scope of this HHERA. Such workers are generally not evaluated in a HHRA, but rather, would be covered under applicable NS Health and Safety Regulations. In addition, it is recommended and expected that appropriate PPE

(personal protective equipment) be worn by workers when on the site, and that applicable NS occupational health and safety regulations and policies be complied with at all times. Although site worker receptors were not evaluated in the HHRA, site soil data (all depths) were reviewed to aid in the preparation of site-specific health and safety plans for Dillon personnel conducting field work on the site. Key parameters and assumptions for the receptors evaluated in the HHRA are described in Table 4-1, below:

Table 4-1: Summary of Parameters and Assumptions for Human Receptors

| Parameter | Value | Reference |
|---|---|----------------------|
| Age (years) | Toddler: 6 months to <5 years Child: 5 years to <12 years Teen: 12 years to <20 years Adult: ≥20 years | Health Canada, 2021a |
| Body Weight (kg) | Toddler: 16.5 Child: 32.9 Teen: 59.7 Adult: 70.7 | Health Canada, 2021a |
| Inhalation Rate (m ³ /day) | Toddler: 8.3 Child: 14.5 Teen: 15.6 Adult: 16.6 | Health Canada, 2021a |
| Soil Ingestion Rate (mg/day) | Toddler: 80 Child: 20 Teen: 20 Adult: 20 | Health Canada, 2021a |
| Hand Surface Area (m ²) | Toddler: 0.043 Child: 0.059 Teen: 0.08 Adult: 0.089 | Health Canada, 2021a |
| Skin Surface Area (upper and lower arms and legs) (m ²) | Toddler: 0.258 Child: 0.455 Teen: 0.720 Adult: 0.822 | Health Canada, 2021a |
| Hand Soil Adherence Factor (kg/cm ² -event) | Toddler: 1 x 10 ⁻⁷ Child: 1 x 10 ⁻⁷ Teen: 1 x 10 ⁻⁷ Adult: 1 x 10 ⁻⁷ | Health Canada, 2021a |
| Skin Soil Adherence Factor (kg/cm ² – event) | Toddler: 1 x 10 ⁻⁸ Child: 1 x 10 ⁻⁸ Teen: 1 x 10 ⁻⁸ Adult: 1 x 10 ⁻⁸ | Health Canada, 2021a |

| Parameter | Value | Reference |
|---|---|----------------------|
| Outdoor Dust Level from Soil (g/m ³) ¹ | 0.00025 | Health Canada, 2021a |
| Time Spent On Site | <p><i>Outdoor site visitor: 2 hours/day, 1 day/week, for 6 months/year (assumed May to October).</i></p> <p>Note: For oral and dermal soil contact exposure pathways (described in Section 4.2.3), exposures are considered event driven, and as such, the hours/day term does not apply in the exposure and risk modelling. Inhalation exposures are considered time driven where the hours/day is included.</p> | Assumed |

Notes:

1. The outdoor dust level assumes vehicular traffic on unpaved roads.

4.2.3

Selection of Exposure Pathways and Scenarios

People can come into contact with chemicals in a variety of ways, depending on their daily activities and their land use patterns. 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, groundwater, and/or sediments; and
- **Dermal absorption** (uptake through the skin) from direct skin contact with gases/vapours, soils/dusts/sediments, water, and other materials.

The potential for adverse health effects to occur as a result of exposure to chemicals, in any medium, is directly related to the exposure pathways. If there is no possible pathway of exposure to a chemical, regardless of its toxic potency or concentration within a given medium, there is no potential for the development of adverse health effects from that chemical.

Exposure scenarios are based on the outcomes of exposure pathway, receptor, and COPC identification/selection at a site, and describe how human exposure to site contaminants may take

place. Exposure scenarios were selected based on the information previously presented in relation to site characterization, COPC identification, receptor selection, exposure pathway selection, and land use.

Based on the information previously presented in relation to site characterization, COPC identification and exposure pathway selection, and assuming that a commercial land use will be applicable in the future, the selected exposure scenario was as follows:

- **Outdoor Site Visitor Scenario:** This scenario assumes that receptors (all age classes except infants) would periodically visit the site. It was assumed that outdoor site visitors would be present on the site for 2 hours per day, 1 day per week for 6 months of the year.

The most likely and relevant human exposure pathways/routes for site-related exposures were considered to be as follows:

- Inhalation of outdoor soils/dusts;
- Ingestion of outdoor soils/dusts;
- Dermal contact with outdoor soils/dusts; and
- Ingestion of groundwater (as potable drinking water).

These exposure pathways were evaluated in the HHRA.

A number of other potential exposure pathways for human receptors were considered, but were ultimately excluded, as briefly summarized in Table 4-2, below. HHRA commonly exclude pathways that are not relevant to the site under investigation or that lack sufficient data to enable their evaluation with a reasonable degree of confidence and/or accuracy.

Table 4-2: Summary of Selected and Excluded Exposure Pathways for the Lake Enon Former Mill Site

| Pathway | Selected or Excluded? | Rationale |
|---|-----------------------|--|
| Inhalation of outdoor soils/dusts | Selected | COPCs have been identified in soils on site. People may be exposed to COPC in surface soils while on site. |
| Dermal Contact with outdoor soils/dusts | Selected | |
| Ingestion of outdoor soils/dusts | Selected | |
| Groundwater ingestion (as drinking water) | Selected | While there are currently no potable groundwater wells on site, some properties close to the site are zoned as residential, meaning that potable wells could potentially be constructed at these locations in the future. Given this, site groundwater was conservatively assumed to be potable. |

| Pathway | Selected or Excluded? | Rationale |
|---|-----------------------|--|
| Ingestion and dermal contact of surface water/sediments | Excluded | <p>While there is surface water present on the site within the former settling/tailings ponds, and within some small site watercourses, this water would not be used for drinking. People would also not swim in the water (e.g., the large tailings pond in AEC 6 has steep slopes on two sides and some parts of it would be difficult to reach; site watercourses are shallow and narrow). If someone accidentally fell into one of the settling / tailings ponds, or watercourses, exposures to COPCs in surface water and sediments would be transient, of very short duration and insignificant.</p> <p>There is the potential for some off-site COPC exposures to human receptors via surface water and sediments in Lake Enon. Cottages are present on the lake, and there are some residential properties along Loch Lomond Road. The nearest residence/cottage appears to be at least 100 m from the site boundary. Swimming could occur in the lake. Surface water samples collected to date from Lake Enon did not result in the identification of any COPCs; see Section 4.2.4.2), but several sediment samples collected from Lake Enon exceeded the CCME residential soil guideline for lead (140 mg/kg). Given this finding, near shore sediment samples may also be high in lead; however, the evaluation of near-shore sediment concentrations in Lake Enon (or other off-site water bodies), was beyond the scope of this HHERA and as such, this pathway was excluded.</p> |
| Dermal contact with groundwater | Excluded | Dermal contact with site groundwater is not expected to occur. If dermal contact with groundwater did occur, it would be transient and of short duration in nature, such that exposure to substances present in groundwater would be negligible. |
| Inhalation of vapours in indoor air | Excluded | There are no habitable or occupied buildings on the site. The construction of buildings on site is not anticipated to occur in the future. |
| Indoor dust ingestion, dermal contact, and ingestion | Excluded | |
| Ingestion of country foods | Excluded | <p>There is no indication that the site is used for country food harvesting (including plants, berries, and local fish and game species). Berry species observed on the site were limited and of low abundance.</p> <p>In addition, the terrain on the site is challenging (e.g., steep areas, wetland/bog areas, dense forest), which would make the site difficult to use for country food harvesting, relative to numerous other sites in the vicinity of the subject site.</p> |

| Pathway | Selected or Excluded? | Rationale |
|--|-----------------------|--|
| Ingestion of agricultural foods (crops, livestock) | Excluded | The site has never been and is unlikely to ever be used for agricultural purposes. Soil and terrain conditions would make any attempt to use the site for agricultural purposes extremely challenging. |
| Outdoor air inhalation | Selected | <p>Though not a site-related exposure pathway, lead (one of the key COPCs) is present in outdoor ambient air at low levels. The status of lead as a non-threshold non-carcinogen with a low toxicity reference value results in the need to consider key non-site sources and pathways of lead exposure in order to enable reasonably thorough exposure and risk estimation, and exposure pathway attribution, and to also enable the development of risk management objectives (such as media-specific site-specific target levels or SSTLs) for a given site that reflect total lead exposures and lead exposure pathway attribution for both site and non-site pathways.</p> <p>When taking this approach for lead, it is also important to take the same approach for other COPCs being assessed (where data are available to enable such an assessment approach), in order to ensure consistency in the HHRA.</p> <p>Thus, the inhalation of typical rural ambient outdoor air COPC concentrations was evaluated in the HHRA.</p> |
| Market basket (grocery) food ingestion | Selected | <p>Though not a site-related exposure pathway, lead (one of the key COPCs) is present in many typical grocery store-bought food items at concentrations that can be elevated (depending on the food item and where it originates from). For the reasons noted above for the air pathway, there is a need to consider key non-site sources and pathways of lead exposure.</p> <p>When taking this approach for lead, it is also important to take the same approach for other COPCs being assessed (where data are available to enable such an assessment approach), in order to ensure consistency in the HHRA.</p> <p>Thus, the ingestion of COPCs from consuming common grocery store food items was evaluated in the HHRA.</p> |

4.2.4

Identification of COPCs

It is common practice in HHRAs to limit the number of chemicals evaluated to those that represent the greatest potential concern to people that may be present in the area under consideration. This is done

because it is often impractical in terms of time and cost to conduct a risk assessment for every chemical that has been found at measurable concentrations in a particular area. In addition, the concentrations of many chemicals associated with a particular site may be similar to chemical concentrations found naturally in the area rather than the result of predicted, current or former anthropogenic activities at the site. It is also preferable to comprehensively evaluate a smaller number of chemicals which represent the greatest potential concern, than it is to conduct a less detailed risk assessment on a larger number of chemicals that are of lesser potential concern.

This step of the HHRA identifies chemicals that have the potential to pose a risk to human health. As per Health Canada (2021a) guidance, COPCs are defined as follows:

- Chemicals for which the maximum on-site concentration exceeds appropriate human health-based environmental quality criteria (e.g., human health-based guidelines or criteria); AND
- Chemicals for which the maximum on-site concentration exceeds local or regional background concentrations; OR
- Chemicals for which no human health-based criteria or background data exist.

The identification of COPCs involved a sequential step-wise process, as follows:

1. Comparison of site soil chemistry data maxima to regulatory human health-based soil quality benchmarks (guidelines) and background data);
2. Chemicals with a maximum concentration that exceeded the applicable human health-based benchmark and were above background concentrations, were identified as initial COPCs. Chemicals for which no health-based benchmark was available were also identified as initial COPCs;
3. These initial COPCs were further evaluated by considering the frequency of detection, and the frequency and degree of exceedance of site soil chemistry data over human health-based soil quality benchmarks; and
4. Further considerations (where/if deemed necessary and relevant), such as: essential nutrient status of certain inorganic elements, ubiquity of certain elements in soils, statistical relationships, local geology, spatial distribution patterns of soil chemical concentrations, as well as environmental fate and behaviour properties (including the potential for a substance to bioaccumulate and biomagnify), and the likely speciation (chemical forms) in site media.

The COPC identification process excluded data for general chemistry and physical properties of environmental media (e.g., chloride, electrical conductivity, sodium absorption ratio, etc.). While such parameters can inform on soil and water quality (where applicable), and may influence or modify the bioavailability, mobility and toxicity of certain COPCs, they are not typically considered directly in the COPC identification step of a HHRA. Also, many of these parameters lack toxicity or human health-based benchmarks that media concentrations can be compared against. As such, these types of parameters were not considered further in the HHRA.

If site media concentrations are less than the applicable guidelines, then the likelihood of potential human health effects related to that chemical can be considered negligible. However, if site media

concentrations exceed these guidelines, it does not necessarily mean that unacceptable risk exists; it is simply an indicator that further assessment is required. Guidelines are intentionally derived by regulatory agencies to be conservative and protective.

The goal of COPC selection is to identify those chemicals measured in site media that may pose a potential risk to human health. Based on the site characterization information, the anticipated future land use of the site, and consideration of the physical-chemical and environmental fate and behaviour properties of the chemicals present in site media, the human exposure media of concern are soil, surface water, and groundwater.

Metals in site soil were assessed (screened) as part of the initial problem formulation completed by Dillon (Dillon, 2023a); however, the focus of the previous problem formulation was on metals in surface soils only, and the additional site data collected as part of the supplemental sampling program (**Appendix C**) were not included in the original screening. As such, the COPC screening was updated for the current HHRA.

The site data and benchmarks (guidelines) used in the COPC screening process are described and discussed in the following sections, and in **Appendix H**. All analytical data used in the COPC screening, along with applicable benchmarks and background data are provided in **Appendix H**. The COPC screening results are discussed in the following sections.

4.2.4.1

COPCs in Site Soil

Determining Usability of Soil Data

For HHRAs, surface soils can be represented by soil depths ranging from 0.05 m bgs to 1.5 m bgs; however, the soil depth that is considered to contribute the greatest to incidental soil exposures via direct contact is typically the top 5 cm to 10 cm (provided the soils are not tilled, excavated, subjected to gardening) (Health Canada, 2021a).

For the screening of COPCs in soil, it is assumed that a soil depth of <0.3 m bgs represents surface soil that is biologically accessible for humans (i.e., soil at depths greater than 0.3 m bgs are generally not available for direct contact or ingestion by humans). Therefore, only soil samples with depths of <0.3 m bgs were used to identify COPCs in soil for the site visitor.

All relevant soil data collected from within the study boundary and local background areas were considered in the surface soil COPC screening.

Comparison of Site Soil Data to Regulatory Human Health-Based Benchmarks

Part of the COPC identification process involved comparing site soil data to available benchmarks that are protective of human health. Maximum measured soil chemical concentrations were compared to

applicable regulatory benchmarks that were derived specifically for the protection of human health as per Health Canada (2021a) guidance (see **Appendix H**).

The specific human health-based soil quality benchmarks that were considered in the COPC identification process are presented below. As the study area is located on Nova Scotia land, NS CSR guidelines apply and the NS PSS for commercial soils were preferentially used.

It is important to recognize that all human health-based benchmarks are derived by their respective regulatory agencies to be intentionally conservative and protective. Thus, exceedance of these benchmarks does not imply there is a risk of adverse health effects; rather, it suggests that further evaluation of those chemicals is warranted.

Given the available soil chemistry information for the site and considering the operable exposure pathways that were selected for the HHRA, soil quality benchmarks that were developed to be protective of direct soil contact exposure pathways (ingestion, dermal contact, and soil/dust inhalation), were considered to be the most relevant and applicable for use. Where applicable, the applied soil quality benchmarks were for a non-potable groundwater use condition and were the lower of benchmarks derived for a fine or coarse soil condition (as the site has both coarse and fine soil types). The non-potable groundwater condition benchmarks were selected as measured site groundwater data are available, and these data were screened separately from soil.

The study area has a current commercial land use designation, which was assumed not to change in the future. As such, commercial land use soil quality guidelines were used in the screening.

Nova Scotia assumes an acceptable target cancer risk level of 1 in 100,000 ($1E^{-5}$), similar to most federal departments and several other provinces. Thus, all soil quality benchmarks (guidelines) used for carcinogens were set to correspond to the $1E^{-5}$ target cancer risk level. The specific guideline sources used in the COPC screening are briefly described below.

Nova Scotia Pathway Specific Standards (NS PSS)

The Nova Scotia Tier II PSS for Soil (2022) - commercial land use; protective of direct soil contact/ingestion; were selected for the COPC screening, where available. The PSS are benchmarks (guidelines) developed by source agencies and departments outside of Nova Scotia such as the CCME soil quality guidelines (SoQGs) developed for the protection of human health (SoQG_{HH}); Alberta Environment and Parks (AEP) guidelines for human health; Ontario Ministry of the Environment, Conservation and Parks [MECP] soil component values; US EPA Regional Screening Levels [RSLs]). The primary guidelines adopted by NSECC, and thus applied herein, are provided below. The original guideline sources were reviewed to ensure the NS PSS guideline referenced the most recent applicable guideline from the original source. The specific source of the guidelines used in the COPC screening are identified in the human health screening tables (**Tables H-1 and H-2, Appendix H**).

Canadian Council of Ministers of the Environment (CCME)

CCME SoQG_{HH} for a commercial land use (lower of coarse or fine-grained soils, where applicable) were then preferentially applied. The direct soil contact guidelines were selected. Where the CCME soil guidelines defer to the older CCME (1991) soil remediation criteria (which are not identified as being either human or ecological health-based), the benchmarks from the next preferred jurisdiction were selected rather than using the outdated CCME (1991) value(s), where possible. If no other guideline was available from another jurisdiction, the CCME (1991) value was used. The most up to date CCME guidelines were selected from on the online CCME summary tables and associated fact sheets (<http://sts.ccme.ca/en/index.html>; searched June 2023).

Atlantic Risk-Based Corrective Action (RBCA)

The Atlantic RBCA (2022) soil quality guidelines for petroleum hydrocarbons for commercial sites were used for PHCs where applicable.

CCME Canada Wide Standards for PHCs in Soils (CCME, 2008)

The Canada Wide Standards (CWS) for PHCs for commercial land use and lower of fine and coarse-grained soils applicable for human health were selected for comparison purposes for PHC F1 to F4 (CCME, 2008). Tier 1 levels protective of direct soil contact were selected. Concentrations were also evaluated against the CWS Management Limits, which are Tier 1 levels that incorporate policy-based considerations to address various scientific, technical, and socioeconomic factors other than chronic toxicity. Details on the basis and derivation of the CWS human health SoQGs are provided in CCME (2008).

Alberta Environment and Parks (AEP, 2022)

The direct soil contact, human health-based soil guidelines for commercial land use, and lower of fine or coarse-grained soils, were applied. These AEP guidelines are developed in a very similar manner to, and are protective of, the same human receptor types as the CCME SoQG_{HH}. Details on soil guideline basis and derivation are provided in AEP (2022).

Ontario Ministry of the Environment, Conservation, and Parks (MECP, 2011)

The MECP human health-based soil component values for soil contact (i.e., soil component values for S1 Risk) were applied. The specific soil component values that were used are from Table 3 in MECP (2011), for conditions of non-potable water, commercial land use, and lower of fine and coarse-textured soils. The derivation of these values is described in MECP (2011).

US Environmental Protection Agency Regional Screening Levels (RSL) (2023)

The US EPA RSLs for residential soil (summary table; May 2023) were applied. RSLs are divided by 5 for non-carcinogens to be consistent with CCME SoQGs, which typically assume a soil allocation factor or target HQ of 0.2 in their derivation (RSLs assume a HQ of 1.0). Details on the basis and derivation of these benchmarks is provided at <https://www.epa.gov/risk/regional-screening-levels-rsls-generic-tables>.

4.2.4.2

Results of the Human Health-Based COPC Screening of Site Soil

COPC screening was conducted using site surface soil data (which is most relevant for the site visitor receptor and exposure scenario) and was also conducted using site soil data from all depths to account for potential future site works where subsoil could be excavated/graded and placed at the soil surface. The comparison of the maximum measured concentrations of chemicals in the site soil at all depths to the selected $SoQG_{HH}$ are presented in **Table 4-3** and the comparison of maximum measured concentrations of chemicals in site surface soil to the selected $SoQG_{HH}$ are presented in **Table 4-4**. Additional details are provided in **Tables H-1 and H-2, Appendix H. Figure 3, Appendix A** shows the soil sampling locations. **Figure 4, Appendix A** shows the exceedances of human health-based guidelines for soils at all soil depths within the study boundary. **Figure 5 to Figure 7, Appendix A** show the locations of the human health guideline exceedances of surface soils within the study boundary. A lead contour map for surface soil is provided in **Figure 8, Appendix A**.

Chemicals that were not detected in any of the analyzed soil samples and were not associated with current and historical activities at the site were excluded from further assessment. For metals, this included antimony and boron. A check was conducted to confirm that chemicals that were not detected in any of the soil samples had detection limits that were greater than applicable benchmarks.

For PAHs, PHCs, VOCs, sVOCs, and phenolics contaminant groups, all analytes were non-detectable with the exception of those listed below **Table 4-3** (see **Appendix E** for soil data for these analytes).

Table 4-3: Human Health Screening for COPCs in Study Area Full Depth Soils, Direct Contact Pathway

| Chemical | Maximum Measured Concentration (mg/kg) | Background Maximum Concentration (mg/kg) | Human Health Screening Level (SoQG _{HH}) (mg/kg) | Exceeds Guideline and Background? (Y/N) ^e |
|--------------------------|--|--|--|--|
| Metals | | | | |
| Aluminium | 31,000 | 38,000 | 15,400 ^c | N |
| Arsenic | 15 | 6.2 | 31 ^a | N |
| Barium | 1,500 | 1100 | 10,000 ^a | N |
| Beryllium | 2 | 1.6 | 110 ^a | N |
| Bismuth | 2.9 | <2.0 | NGA | NA |
| Cadmium | 25 | 2.6 | 49 ^a | N |
| Chromium (Total, III+VI) | 34 | 21 | 630 ^a | N |
| Cobalt | 270 | 13 | 22 ^b | Y |
| Copper | 120 | 42 | 4,000 ^a | N |
| Iron | 100,000 | 31,000 | 11,000 ^c | Y |
| Lead | 30,000 | 170 | 260 ^a | Y |
| Manganese | 8,900 | 5,200 | 360 ^c | Y |
| Mercury | 0.39 | 0.56 | 24 ^a | N |
| Molybdenum | 12 | 2.7 | 110 ^b | N |
| Nickel | 69 | 18 | 310 ^a | N |
| Selenium | 3.5 | 14 | 125 ^a | N |
| Rubidium | 28 | 29 | NGA | NA |
| Lithium | 35 | 41 | 32 ^c | N |
| Silver | 4.3 | 1.3 | 77 ^b | N |
| Strontium | 27,000 | 470 | 9,400 ^c | Y |
| Thallium | 0.52 | 0.42 | 1 ^a | N |
| Tin | 5.5 | 1.7 | 9,400 ^c | N |
| Uranium | 11 | 3.9 | 33 ^a | N |
| Vanadium | 73 | 43 | 39 ^b | Y |
| Zinc | 3,600 | 500 | 16,000 ^a | N |

| Chemical | Maximum Measured Concentration (mg/kg) | Background Maximum Concentration (mg/kg) | Human Health Screening Level (SoQG _{HH}) (mg/kg) | Exceeds Guideline and Background? (Y/N) ^e |
|---|--|--|--|--|
| Polycyclic Aromatic Hydrocarbons | | | | |
| 1- and 2-Methylnaphthalene ^f | 0.1 | - | 72 ^{b,d} | N |
| Fluoranthene ^f | 0.06 | - | 7.8 ^b | N |
| Phenanthrene ^f | 0.031 | - | 41,000 ^{b,k} | N |
| Petroleum Hydrocarbons (PHCs) | | | | |
| Modified TPH ^g | 2600 | - | 21,000 ^e | N |
| PHC F2 (>C10-C16) ^h | 17 | - | 10,000 ⁱ | N |
| PHC F3 (>C16-C34) ^h | 950 | - | 23,000 ⁱ | N |
| PHC F4 (>C34-C50) ^h | 519 | - | >30,000 ^{l,j} | N |

Notes:

N = 107 for all metals except lead where N = 118

NGA = no guideline available; NA = not applicable; Y = yes; N = no.

Bold values indicate exceedances over the guideline and background.

Italicized values indicate exceedances over the guideline, but not background.

No background surface water concentrations in area wetlands were available, and as such, background is noted as “-” in the table above.

- a. Guideline is from CCME Factsheets. SoQG_{HH} (soil quality guideline human health); commercial land use; lower of direct contact pathways; 10-5 risk level (1 in 100,000) where applicable.
- b. MECP (2011) Table 3 Soil Components - Full Depth, Non-Potable Water - Institutional land use; Lower of fine or coarse-grained soils - Soil contact (human); S1 Risk soil component.
- c. US EPA Regional Screening Level (RSL) - Resident Soil RSL May, 2023; lowest of ingestion, dermal and inhalation pathways; divided by 5 to adjust for soil pathway only as per Health Canada, 2021.
- d. No guideline is available for 1- or 2-methylnaphthalene as individual compounds; the guideline presented is for the sum of 1 & 2-methylnaphthalene, and the maximum concentration presented is the sum of the maximum concentration of each.
- e. NS PSS (2021) for Modified TPH, lube oil fraction from Atlantic Risk Based Corrective Action (RBCA) (Atlantic PIRI, 2022). Lube oil fraction applied as all samples showed resemblance of lube fraction.
- f. N = 23
- g. N = 8
- h. N = 1; Of the 6 samples analyzed for BTEX/TPH, one sample, SSSS12, did not reach baseline. Therefore, F4 analysis was requested which includes the F2-F4 fractions. As such, there is only one sample result for PHC F2 to F4.
- i. Canada Wide Standard (CWS; CCME, 2008) for direct contact (ingestion and dermal contact), commercial land use, lower of coarse and fine-grained soils.
- j. PHC F4 guideline was >RES. The calculated guideline exceeded 30,000 mg/kg and solubility limit for PHC fraction. for petroleum hydrocarbons (CCME, 2008)
- k. No guideline is available for phenanthrene; however, given that its structure is similar to anthracene (both 3 rings), the anthracene guideline was applied.

Table 4-4: Human Health Screening for COPCs in Study Area Surface Soils, Direct Contact Pathway

| Chemical | Maximum Measured Concentration (mg/kg) | Background Maximum Concentration (mg/kg) | Human Health Screening Level (mg/kg) | Exceeds Guideline and Background? (Y/N) ^e |
|--------------------------|--|--|--------------------------------------|--|
| <i>Aluminium</i> | 26,000 | 38,000 | 15,400 ^c | N |
| Arsenic | 69 | 6.2 | 31 ^a | Y |
| Barium | 1,500 | 1100 | 10,000 ^a | N |
| Beryllium | 2 | 1.6 | 110 ^a | N |
| Bismuth | 2.9 | <2.0 | NGA | NA |
| Cadmium | 25 | 2.6 | 49 ^a | N |
| Chromium (Total, III+VI) | 34 | 21 | 630 ^a | N |
| Cobalt | 270 | 13 | 22 ^b | Y |
| Copper | 120 | 42 | 4,000 ^a | N |
| Iron | 100,000 | 31,000 | 11,000 ^c | Y |
| Lead | 30,000 | 170 | 260 ^a | Y |
| Manganese | 8,900 | 5,200 | 360 ^c | Y |
| Mercury | 0.35 | 0.56 | 24 ^a | N |
| Molybdenum | 12 | 2.7 | 110 ^b | N |
| Nickel | 69 | 18 | 310 ^a | N |
| Selenium | 3.5 | 14 | 125 ^a | N |
| Rubidium | 28 | 29 | NGA | NA |
| Lithium | 33 | 41 | 32 ^c | N |
| Silver | 2.5 | 1.3 | 77 ^b | N |
| Strontium | 27,000 | 470 | 9,400 ^c | Y |
| Thallium | 0.52 | 0.42 | 1 ^a | N |
| Tin | 2.7 | 1.7 | 9,400 ^c | N |
| Uranium | 11 | 3.9 | 33 ^a | N |
| Vanadium | 73 | 43 | 39 ^b | Y |
| Zinc | 1,500 | 500 | 16,000 ^a | N |

Notes:

N = 88 for all metals except lead where N = 99.

NGA = no guideline available; NA = not applicable; Y = yes; N = no.

Bold values indicate exceedances over the guideline and background.

Italicized values indicate exceedances over the guideline, but not background.

- Guideline is from CCME Factsheets. SoQG_{HH} (soil quality guideline human health); commercial land use; lower of direct contact pathways; 10⁻⁵ risk level (1 in 100,000) where applicable.
- MECP (2011) Table 3 Soil Components - Full Depth, Non-Potable Water - Institutional land use; Lower of fine or coarse-grained soils - Soil contact (human); S1 Risk soil component.
- US EPA Regional Screening Level (RSL) - Resident Soil RSL May, 2023; lowest of ingestion, dermal and inhalation pathways; divided by 5 to adjust for soil pathway only as per Health Canada, 2021.

Substances with a maximum concentration which exceeded the applicable guideline are shown in **Table 4-3** and **Table 4-4**. These chemicals were also compared to the maximum background soil concentration. Background soil data were collected from within the assessment properties but outside of the study boundary (see **Figure 3, Appendix A**).

All aluminium samples that exceeded the guideline (**Tables 4-3** and **4-4**) were below the maximum local background concentration of 38,000 mg/kg. For this reason, aluminium in site soil was considered to be background related and was not carried forward as an initial COPC.

Iron exceeded the applicable guideline of 11,000 mg/kg (US EPA RSL, 2023) in 90 of 107 full depth samples and 78 of 90 surface soil samples by up to 9.1-fold. The maximum local background concentration for the site was 31,000 mg/kg. The background maximum was exceeded in only 5 of 88 samples (for both surface and full depth soils), by up to a maximum of approximately 3.2-fold in sample TP-22 (depth of 0.15 to 1.0 m). A sample collected at this same location (TP22) at a depth of 0 to 0.15 m had a concentration of 49,000 mg/kg which exceeded the maximum background by approximately 2.8-fold. The remaining soil samples were all within the background range. Soil metals data were also compared to the Nova Scotia Maximum Concentration Background Levels for the Highlands Soil Zone of 52,000 mg/kg, as presented in the Environment Canada (EC) Background Soil Database (EC, 2022). The maximum iron concentration at TP22 for the 0.15 m to 1.0 m depth sample exceeded the Highlands Soil Zone background concentration by less than 2-fold, while the surface sample collected at TP22 (0 to 0.15 m) of 49,000 mg/kg and all other samples were below the Highland Soil Zone background iron concentration. Given the high background concentrations of iron and low frequency of exceedance over the background maximum, iron in site soil was considered to be background related and was not carried forward for further consideration.

The maximum manganese concentration in soil within the study boundary was 8,900 mg/kg. Manganese exceeded the applicable guideline of 360 mg/kg in 61 of 62 full depth soil samples and 86 of 90 surface soil samples by up to 25-fold (sample TH57: 0-0.15 m). The maximum local background concentration for the site was 5,200 mg/kg. The maximum background concentration was only exceeded in six of 90 surface soil samples (TP-27, TP2, MW5 SS1, TH42 5500, TH57 8900, TH60 8700), and the remaining samples were all within the background range. The maximum site manganese concentration of 8,900 mg/kg at TH57 (0 to 0.15 m) exceeded the Highland Soil Zone background manganese concentration of 4,340 mg/kg by approximately 2-fold. Manganese is elevated in many areas of Nova Scotia and manganese ores have been mined on Cape Breton Island. The Loch Lomond area of Cape Breton Island, which is in the vicinity of the subject property, was indicated to be one of the principal manganese areas in NS (Bishop and Wright, 1974). Given the high background concentrations of manganese in the region and the low frequency of exceedance of the background maximum, manganese was not carried forward as an initial COPC.

Based on the number of site soil samples that exceeded the applicable soil quality guideline (provided in brackets) and given consideration of the local background data, the following chemicals were selected as initial COPCs:

Initial COPCs – Direct Soil Contact: Full Depth of Soil

- Cobalt (3 of 107);
- Lead (57 of 118);
- Strontium (13 of 107); and
- Vanadium (5 of 107).

Initial COPCs – Direct Soil Contact: Surface Soil

- Arsenic (1 of 90);
- Cobalt (3 of 90);
- Lead (52 of 101);
- Strontium (12 of 90); and
- Vanadium (4 of 90).

Further Examination of Initial COPCs

Arsenic exceeded the applicable guideline of 31 mg/kg in only 1 of 88 surface samples (TH77 collected in September 2023) at a concentration of 69 mg/kg (2.2-fold over the guideline). This sample was collected at 0 to 0.15 m bgs. Given the low frequency and degree of exceedance, arsenic was not carried forward for further consideration.

Cobalt exceeded the applicable guideline in 3 of 107 (2.8%) full depth samples and 3 of 90 (3.3%) surface soil samples, with the maximum soil concentration within the study boundary of 270 mg/kg exceeding the guideline of 22 mg/kg by 12.2-fold (TP22, depth of 0.15 to 1.0 m). A sample collected at this same location (TP22) but at a depth of 0 to 0.15 m had a cobalt concentration of 120 mg/kg, which exceeded the guideline by 5.5-fold. One other sample exceeded the guideline by 1.8-fold (TH68; 39 mg/kg). These samples also exceeded the maximum background concentration of 13 mg/kg. The UCLM95 concentration of cobalt in soil within the study boundary was calculated to be 19 mg/kg, which is below the cobalt human health-based soil quality guideline. Given the low frequency of exceedance, cobalt was not carried forward for further consideration.

Lead exceeded the applicable guideline of 260 mg/kg in 57 of 118 full depth samples and 52 of 101 surface samples, by as much as 115-fold. The background maximum for lead was 170 mg/kg. Given the high frequency and degree of exceedance, lead was carried forward for further consideration.

Strontium exceeded the applicable guideline of 9,400 mg/kg in 13 of 107 full depth samples and 12 of 90 surface soil samples by up to 2.9-fold. Given the relatively high frequency and degree of exceedance, strontium was carried forward for further consideration.

Vanadium exceeded the applicable guideline of 39 mg/kg in 14 of 107 full depth samples and 11 of 90 surface soil samples by less than two-fold. One sample exceedance occurred at TP22 at a depth of 0.15 to 1.0 m (57 mg/kg); however, a sample from this same location collected at 0 to 0.15 m (36 mg/kg) did not exceed the guideline. One other sample exceeded the guideline by 1.3-fold (TH68; 52 mg/kg). The maximum soil concentration of vanadium within the study boundary was 73 mg/kg (TH46, 0 to 0.15 m), while the maximum for the background was 43 mg/kg. When background soil concentrations are considered, only 4 of 90 (4.4%) surface soil samples and 5 of 107 (4.7%) full depth soil samples exceed the guideline and the maximum background concentration. The UCLM95 concentration of vanadium in soil within the site boundary was calculated to be 28.8 mg/kg, which is below the human health-based soil quality guideline. Given the elevated background concentration of vanadium, and the exceedances being less than 2-fold over the guideline, vanadium was not carried forward for further consideration.

Based on the available study area soil data, and the COPC identification steps and considerations described above, the following COPCs were carried forward as final COPCs for the Site Visitor exposure scenario:

Final COPCs – Direct Soil Contact (Surface Soils)

- Lead; and
- Strontium.

4.2.4.3

COPCs in Site Surface Water

No surface waters occur within the study boundary with the exception of surface water that is in the settling/tailings ponds, which would not be relevant for human consumption. While someone could possibly accidentally fall into one of these ponds, the resulting exposures would be infrequent, transient, and likely insignificant. Thus, potential exposures to COPCs in the surface water of site tailings and settling ponds were not evaluated within the HHRA.

However, Lake Enon is adjacent to the site, and water from this off-site lake could potentially be used for drinking water purposes by local residents with homes or cottages on or near the lake. Thus, surface water samples collected from Lake Enon were analyzed for metals and analytical results were compared to drinking water guidelines protective of human health.

Although the tailing and settling ponds are not relevant for human consumption (this pathway of exposure was excluded in **Section 4.2.3**), samples from these site water features were conservatively included in the drinking water guideline comparison. The one sample and one duplicate sample collected at the discharge point into Lake Enon (i.e., SW-D) were also included in the drinking water guideline comparison.

BTEX and PHCs, PAHs, VOCs, sVOCs, energetics, anilines, phenolics, and phthalates were not analyzed for in Lake Enon, but were analyzed for (and were generally not detected) in settling/tailings ponds. One

pond sample contained a low concentration of phenanthrene, and two pond samples contained diethylphthalate at a concentration equal to the detection limit. As such, these parameters were not considered further in the drinking water guideline comparison (**Appendix E**).

The benchmarks (guidelines) used to screen the Lake Enon and site surface water data are described below.

Health Canada Drinking Water Quality Guidelines

Health Canada Guidelines for Canadian Drinking Water Quality (September 2022); risk-based Maximum Acceptable Concentrations (MACs) were used where available. If risk-based values did not exist for a given substance, then the next preferred jurisdiction was reviewed. If no risk based values were available, values referenced as ALARA (as low as reasonably achievable) were selected. These guidelines can be found at <https://www.canada.ca/en/health-canada/services/environmental-workplace-health/reports-publications/water-quality/guidelines-canadian-drinking-water-quality-summary-table.html> (searched August, 2023).

US Environmental Protection Agency Regional Screening Levels (RSL) (2023)

The US EPA RSLs for tap water (summary table; May 2023) were applied if no Canadian drinking water quality guideline was available. RSLs are divided by 5 for non-carcinogens to be consistent with Health Canada drinking water quality guidelines, which typically assume a target HQ of 0.2 in their derivation (RSLs assume a HQ of 1.0). Details on the basis and derivation of these benchmarks is provided at: <https://www.epa.gov/risk/regional-screening-levels-rsls-generic-tables>.

Results of the Human Health-Based COPC Screening of Surface Water

The results of the human health screening of Lake Enon, settling/tailings ponds surface water (as well as the discharge sample to Lake Enon) are provided in **Table 4-5** below, and in **Table H-3, Appendix H**. **Figure 3, Appendix A** shows the surface water sampling locations.

The following substances were below their laboratory RDLs (i.e., non-detectable) in all Lake Enon and other surface water samples that were analyzed for these parameters, and were not considered further: antimony, arsenic, beryllium, bismuth, boron, chromium, cobalt, mercury, molybdenum, nickel, silver, thallium, tin, and vanadium. Available drinking water guidelines for these chemicals were greater than the detection limits (**Table H-3, Appendix H**), with the only exception to this being thallium. While the thallium guideline was lower than the detection limit, thallium concentrations were not detected in either the background area or within the site boundary. As such thallium was not considered further.

No drinking water guideline was available for titanium which was not detected in the Lake Enon samples and was only detected in one of the two discharge samples to Lake Enon at 0.003 mg/L, which was only slightly over the detection limit of 0.002 mg/L. As such, titanium was not considered further.

Table 4-5: Human Health Screening for COPCs in Surface Water: Lake Enon, Site Tailings and Settling Ponds and Site Discharge to Lake Enon Samples

| Chemical | Maximum Measured Concentration (mg/L) | Background Maximum Concentration (mg/L) | Human Health Screening Level (mg/L) | Exceeds Guideline and Background? (Y/N) |
|-----------------|--|--|--|--|
| Aluminium | 0.19 | 0.052 | 2.9 ^a | N |
| Barium | 0.43 | 0.082 | 2 ^a | N |
| Cadmium | 0.00011 | 0.000015 | 0.007 ^a | N |
| Copper | 0.0013 | 0.00051 | 2 ^a | N |
| Iron | 0.16 | <0.05 | 0.3 ^{b,c} | N |
| Lead | 0.007 | 0.0018 | 0.005 ^a | Y |
| Manganese | 0.09 | 0.017 | 0.12 ^a | N |
| Selenium | 0.00085 | <0.00050 | 0.05 ^a | N |
| Strontium | 11 | 1.8 | 7 ^a | Y |
| Titanium | 0.003 | <0.0020 | NGA | N |
| Uranium | 0.00085 | <0.00010 | 0.02 ^a | N |
| Zinc | 0.028 | 0.0056 | 1.2 ^{b,c} | N |

Notes:

N = 22

NGA = no guideline available; NA = not applicable; Y = yes; N = no.

Bold values indicate exceedances over the guideline and background.*Italicized* values indicate exceedances over the guideline, but not background.

- ^a Guideline for Canadian Drinking Water Quality guidelines, maximum allowable concentration (MAC), September 2022.
- ^b US EPA Regional Screening Level (RSL) (US EPA, 2023) for tap water, divided by 5 to account for exposures by one pathway only, as per Health Canada, 2021a
- ^c The Health Canada (2022) Guideline for Canadian Drinking Water Quality was based on an aesthetic objective (AO) and no health-based limit was available; September 2022. As such, the health-based US EPA RSL for tap water was used instead.

Surface water samples were below the human health-based drinking water quality guidelines, with the exception of lead in 1 of 17 samples and strontium in 2 of 17 samples (by less than 2-fold). Considering the number of samples collected (N=17) and the local background data, the following chemicals were selected as initial COPCs (# of guideline exceedances provided in brackets) for surface water.

Initial COPCs – Ingestion (Surface Water)

- Lead (1 of 17 samples); and
- Strontium (2 of 17 samples).

Further Examination of Initial COPCs

The maximum lead concentration of 0.007 mg/L (SW-D; December 2022) only slightly exceeded the Health Canada (2022) drinking water quality guideline of 0.005 mg/L by 1.4-fold. The surface water sample collected at this same location in July 2023 was below the guideline. The strontium guideline of 7 mg/L was only slightly exceeded in sample SW-D in both the December 2022 sample (10 mg/L; 1.4-fold)

and the July 2023 sample (11 mg/L; 1.6-fold). All samples with exceedances occurred at the SW-D sample location. This sample location is a discharge pipe at the edge of the site that goes into Lake Enon and would not be an area that would be used to draw water for drinking, or for swimming. Based on the low frequency and degree of exceedances over the drinking water guidelines and given consideration of the sample locations with guideline exceedances, no COPCs were identified in on-site or off-site surface water bodies with respect to human health.

Final COPCs – Ingestion (Surface Water)

- None

4.2.4.4

COPCs in Site Groundwater

Groundwater samples from within the study boundary were analyzed for metals, BTEX and PHCs, PAHs, VOCs, sVOCs, energetics, anilines, phenolics, and phthalates. Analytical results were compared to groundwater benchmarks (guidelines) protective of human health. The benchmarks used to screen the site groundwater data are described below.

Nova Scotia Pathway Specific Standards (NS PSS)

The Nova Scotia Tier II PSS for Groundwater – commercial land use for potable drinking water; for the lower of fine or coarse soils (as both coarse and fine soils were identified within the study boundary) were selected for the COPC screening, where available. The PSS for groundwater is benchmarks developed by source agencies and departments outside of Nova Scotia such as the Health Canada (2022) drinking water quality guidelines, Ontario Ministry of the Environment, Conservation and Parks [MECP] groundwater component values; US EPA Regional Screening Levels [RSLs]). The primary guidelines adopted by NSECC, and thus applied herein, are provided below. The specific source of the guidelines used in the COPC screening are identified in the human health screening tables (**Table H-4, Appendix H**). The NS PSS guidelines were checked with the noted guideline sources to ensure the selected guidelines were up to date.

Health Canada Drinking Water Quality Guidelines

Health Canada Guidelines for Canadian Drinking Water Quality (September 2022); risk-based Maximum Acceptable Concentrations (MACs) were used where available. If risk-based values did not exist for a given substance, then the next preferred jurisdiction was reviewed. If no risk-based values were available, values referenced as ALARA (as low as reasonably achievable) were selected. Guidelines can be found at <https://www.canada.ca/en/health-canada/services/environmental-workplace-health/reports-publications/water-quality/guidelines-canadian-drinking-water-quality-summary-table.html> (searched August, 2023).

Ontario Ministry of the Environment, Conservation, and Parks (MECP, 2011)

The MECP human health-based groundwater component values for potable water (i.e., GW1 values) were applied. The specific groundwater component values that were used are for conditions of potable

water, commercial land use, and were the lower of values for fine or coarse-textured soils. The derivation of these values is described in MECP (2011).

US Environmental Protection Agency Regional Screening Levels (RSL) (2023)

The US EPA RSLs for tap water (summary table; May 2023) were applied if no Canadian drinking water quality guideline was available. RSLs are divided by 5 for non-carcinogens to be consistent with Health Canada drinking water guidelines, which typically assume a target HQ of 0.2 in their derivation (RSLs assume a HQ of 1.0). Details on the basis and derivation of these benchmarks are provided at <https://www.epa.gov/risk/regional-screening-levels-rsls-generic-tables>.

Results of the Human Health-Based COPC Screening of Site Groundwater

The results of the screening of groundwater data collected within the study boundary for human health are provided in **Table 4-6** below, and in **Table H-4, Appendix H. Figure 3, Appendix A** shows the groundwater sampling locations.

The following substances were below their laboratory RDLs (i.e., non-detectable) in all groundwater samples that were analyzed for these parameters and were not considered further (**Table H-4, Appendix H**): beryllium, bismuth, chromium, cyanide, mercury, silver, thallium, tin, titanium, and vanadium. Similarly, BTEX and PHCs, PAHs, VOCs, sVOCs, energetics, anilines, phenolics, and phthalates were not detected in site groundwater samples, with the exception of 1 sample and one duplicate sample which contained a low concentration of 2,4-dimethylphenoldiethylphthalate. No guideline was available for 2,4-dimethylphenoldiethylphthalate; however, this chemical would not be expected to be present in site groundwater as a result of former site activities; thus, it was not considered further.

No drinking water guideline was available for potassium, and the maximum concentration was greater than the background. However, potassium is an essential nutrient, and its absorption, metabolism, distribution and elimination are physiologically or biochemically regulated such that adverse effects would not be expected to occur except in conditions of extremely high exposure. Further, potassium is not associated with any known former or current site activities or structures. As such, potassium was not carried forward for further assessment.

Table 4-6: Human Health Screening for COPCs in Study Area Groundwater (Ingestion)

| Chemical | Maximum Measured Concentration (mg/L) | Background Maximum Concentration (mg/L) | Human Health Screening Level (mg/L) | Exceeds Guideline and Background? (Y/N) |
|-----------|---------------------------------------|---|-------------------------------------|---|
| Aluminium | 0.031 | <0.0050 | 2.9 ^a | N |
| Antimony | 0.0017 | <0.0010 | 0.006 ^a | N |
| Arsenic | 0.0022 | <0.0010 | 0.010 ^a | N |

| Chemical | Maximum Measured Concentration (mg/L) | Background Maximum Concentration (mg/L) | Human Health Screening Level (mg/L) | Exceeds Guideline and Background? (Y/N) |
|------------|---------------------------------------|---|-------------------------------------|---|
| Barium | 0.26 | 0.4 | 2 ^a | N |
| Boron | 0.081 | <0.05 | 5 ^a | N |
| Cadmium | 0.00093 | 0.00035 | 0.007 ^a | N |
| Calcium | 340 | 28 | NR ^a | N |
| Cobalt | 0.0053 | 0.00072 | 0.003 ^c | Y |
| Copper | 0.035 | <0.00050 | 2 ^a | N |
| Iron | 0.21 | <0.05 | 2.8 ^{b,d} | N |
| Lead | 0.0048 | <0.00050 | 0.005 ^a | N |
| Magnesium | 64 | 2.2 | NR ^a | |
| Manganese | 3.2 | 0.4 | 0.120 ^a | Y |
| Molybdenum | 0.26 | 0.007 | 0.070 ^c | N |
| Nickel | 0.46 | <0.0020 | 0.1 ^c | N |
| Potassium | 14 | 3 | NGA | N |
| Selenium | 0.01 | <0.00050 | 0.050 ^a | N |
| Sodium | 14 | 14 | 200 ^{a,e} | N |
| Strontium | 39 | 0.31 | 7 ^a | Y |
| Uranium | 0.012 | 0.00069 | 0.020 ^a | N |
| Zinc | 0.2 | <0.0050 | 1.2 ^b | N |

Notes:

N = 11

NGA = no guideline available; NA = not applicable; Y = yes; N = no; NR = not required.

Bold values indicate exceedances over the guideline and background.*Italicized* values indicate exceedances over the guideline, but not background.

- Guideline for Canadian Drinking Water Quality guidelines, maximum allowable concentration (MAC) where available; for lead and arsenic guideline ALAR or as low as reasonably achievable (September 2022).
- US EPA Regional Screening Level for tap water, divided by 5 to account for exposures by one pathway only, as per Health Canada, 2021a.
- MOECP, 2011 – Groundwater components for potable water scenario (GW1).
- The Health Canada (2022) Guideline for Canadian Drinking Water Quality was based on an aesthetic objective (AO) and no health-based limit was available; September 2022. As such, the health-based US EPA RSL for tap water was used instead.
- No health-based guideline for sodium was identified. The guideline used in the screening was the Health Canada (2012) AO (aesthetic objective) of ≤200 mg/L. Guideline is based on taste so individuals would be unlikely to ingest the water if concentrations were too high.

Based on the groundwater data presented in **Table 4-6** above, and in **Table H-4, Appendix H**, all substances present in groundwater samples collected within the study boundary were below the human health-based groundwater quality guidelines with the exception of: cobalt (3 of 11 samples); manganese (8 of 11 samples), and strontium (10 of 11 samples). Thus, the following chemicals were selected as initial COPCs for groundwater:

Initial COPCs – Groundwater Ingestion

- Cobalt (3 of 11);
- Manganese (8 of 11); and
- Strontium (10 of 11).

Further Examination of Initial COPCs

Of the eleven groundwater samples analyzed for cobalt, two samples and one duplicate sample exceeded the human health-based benchmark of 0.003 mg/L (**Table H-4, Appendix H**). The maximum cobalt concentration of 0.0053 mg/L only slightly exceeded the benchmark by 1.4-fold. Given the low degree of exceedance, and the lack of association of cobalt with former or current site activities and structures, cobalt was not considered to require further assessment.

Manganese in site groundwater exceeded the human health benchmark of 0.120 mg/L in 8 samples up to a maximum of approximately 27-fold (i.e., MW3 at 3.2 mg/L). The background sample (MW5 at 0.4 mg/L) also exceeded the guideline, but to a lesser extent. Note that groundwater in the province frequently exceeds the manganese drinking water guideline of 0.120 mg/L due to naturally elevated concentrations of manganese-rich minerals in both overburden and bedrock. When the Lake Enon area is searched on the Province of Nova Scotia's Relative Risk of Manganese in Water Wells database, the areas within the site boundary are considered a medium risk zone where 5% to 15% of well water samples are expected to naturally exceed the drinking water guideline for manganese (https://fletcher.novascotia.ca/DNRViewer/index.html?viewer=Manganese_Risk). Given this, manganese in site groundwater was assumed to be related to background concentrations and was not carried forward as a COPC.

Strontium exceeded the drinking water guideline in 10 of the 11 site groundwater samples with the maximum concentration at MW5 (39 mg/L) exceeding the applicable groundwater guideline of 7 mg/L by 5.4-fold.

When examining the groundwater exceedances, it is important to consider the location of the samples collected and the depth of groundwater wells. Most of the groundwater from within the study boundary would be expected to flow toward Lake Enon. As such, groundwater would not likely flow towards off-site private wells to the northeastern portion of the site. The closest groundwater well to the off-site properties in this area would be MW8 (7.8 m deep). MW8 is located in one of the on-site soil "hot-spots" for lead (see **Figure 8, Appendix A**). No groundwater drinking water guideline exceedances were noted

at MW8 and lead was not detected in this well (<0.005 mg/L). Potable wells would typically be deeper than 7.8 m; however, when trying to install a deeper well at MW8, it kept collapsing so a deeper sample could not be collected. Other wells where groundwater exceedances were noted were generally 4 to 5 m deep. Given the lack of exceedances at MW8, there is no reason to believe groundwater contamination would extend deeper. Thus, while strontium (and other site-related contaminants) is unlikely to enter drilled potable wells, strontium was conservatively carried forward as a COPC in groundwater, based on its high frequency of drinking water guideline exceedance in the site groundwater samples.

Based on the available data, and the COPC identification steps and considerations described above, the following COPCs were carried forward as final COPCs for human receptors in site groundwater:

Final COPCs – Groundwater Ingestion

- Strontium.

4.2.4.1

Final COPCs in the HHRA

Based on the COPC identification steps and considerations described in the preceding sections, the following COPCs were carried forward for quantitative human health risk assessment:

Soil:

- Lead; and
- Strontium.

Surface Water:

- No COPCs.

Groundwater:

- Strontium.

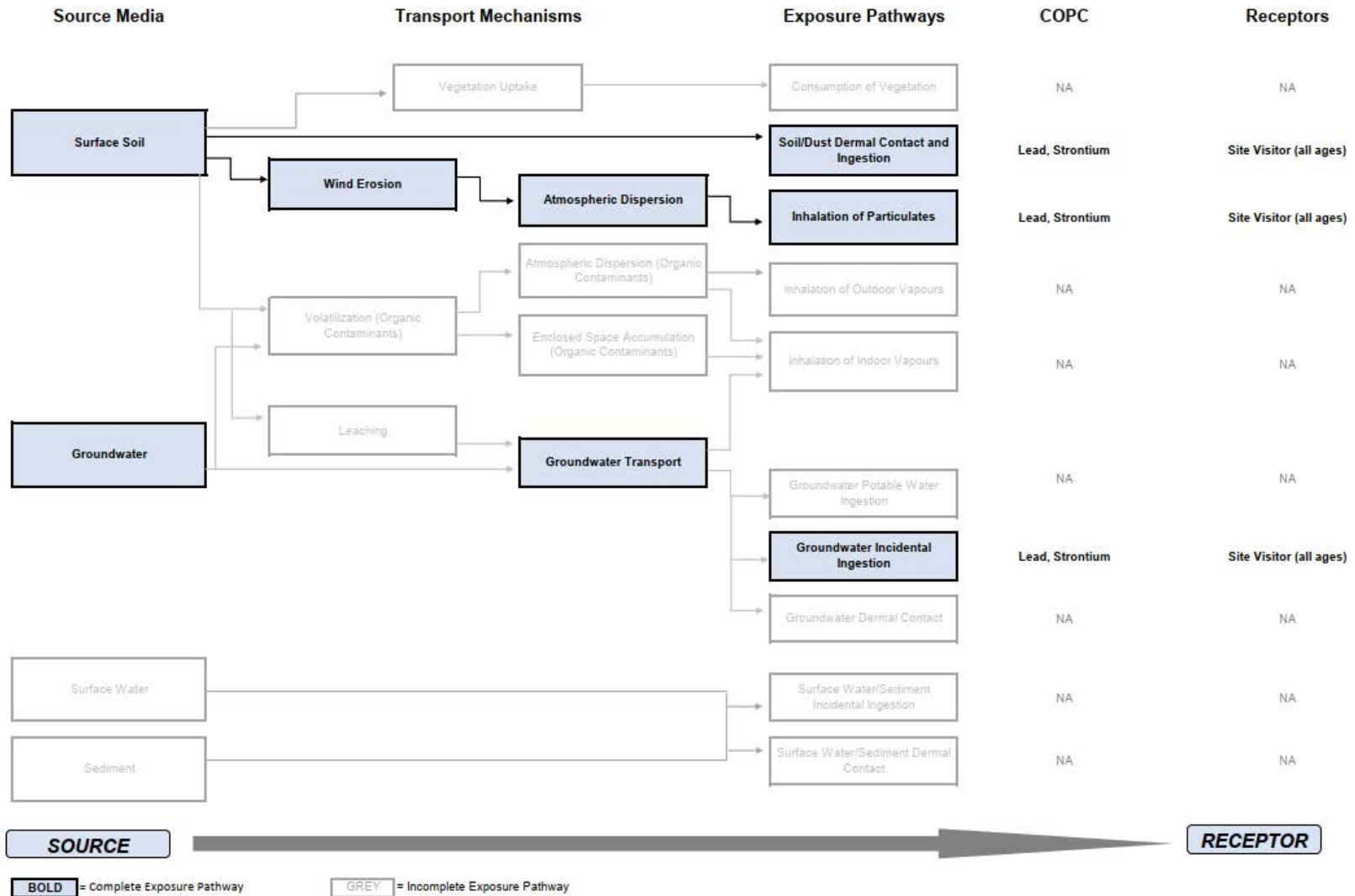
It is noted that if a COPC was identified in one site medium, it was also assessed in all other relevant site media during the HHRA. For example, lead was only identified as a COPC in site soil, but it was also assessed in site groundwater and in site and off-site surface water, where it was conservatively assumed that surface water or groundwater (represented by site groundwater) could potentially be ingested by site visitors as drinking water.

4.2.5

Conceptual Site Model

A CSM provides a description of relevant exposure pathways for the COPCs that have the potential to contribute to human exposures and potential health risks for the selected receptors. The HHRA CSM is presented in **Figure 4-2**. The CSM provides a simplified representation of relevant exposure pathways and routes that link the identified COPCs in site environmental media to the selected human receptors.

Figure 4-2: Conceptual Site Model for Soils for Human Receptors for the Lake Enon Former Mill Site, Enon, Nova Scotia



4.3 Exposure Assessment

The exposure assessment of an HHRA involves estimation of exposure of hypothetical human receptors to each of the COPCs. Exposure assessment involves the estimation of 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 and integrates available data for the chemicals, human receptors, and exposure pathways that were selected during the problem formulation step of the HHRA. The rate of exposure to chemicals, from the various environmental media being evaluated at a given site, 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 at a given site depends on the interactions of a number of parameters, including:

- The concentrations of chemicals in various site environmental media (e.g., air, water, soil, or foods) as determined by the quantities of chemicals entering the site's environmental media from various sources, their fate and persistence in these media, and the normal ambient, or background concentrations that may exist independent of a specific source;
- The various exposure pathways and routes 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 water or soil/dust contact);
- The physiological, behavioural and lifestyle characteristics of human receptors that determine the actual exposures received through interactions with the various exposure pathways (e.g., respiration rate, water intake rate, food intake rate, soils/dusts intake rate, time spent at various activities and in different locations); and
- The various physical, chemical, and biological factors that determine the ability of people to take chemicals into their bodies from the exposure pathways and media (e.g., bioavailability of the chemicals within soil/dust, foods, water, and air).

An exposure assessment can focus on one or more possible pathways. A multimedia exposure assessment approach considers total exposure of human receptors to COPCs through all possible sources or media that can be characterized (i.e., soil, air, dust, water, food, consumer products). Current approaches to the assessment of lead in HHRAs typically necessitate the application of a multimedia and multi-pathway approach where all site and non-site (background) sources and exposure pathways for lead are characterized to the extent possible or practical. This requirement for HHRAs of lead reflects the unique status of lead as a non-threshold, non-carcinogenic substance with a low toxicity reference value. In HHRAs that assess substances in addition to lead (which is common), it is important to maintain a consistent approach in the HHRA; thus, strontium (the other site COPC) was also assessed using a multimedia and multi-pathway HHRA approach.

For the current HHRA, deterministic (or point estimate) exposure analysis was conducted. This is the typical approach to exposure analysis that is taken in most HHRAs and is almost always the approach taken for initial HHRA studies of a given site or study area. Deterministic exposure analysis involves calculating reasonable upper bound point estimate exposures for the receptor-exposure pathway-COPC combinations, under each of the exposure scenarios evaluated. This is a conservative approach that tends to overestimate exposures and risks, but also helps to identify areas where more detailed or advanced approaches (such as further data collection, more detailed statistical analyses, use of probabilistic modelling techniques) could/should be used to refine the various exposure-related assumptions and enable a more site-specific and realistic exposure assessment. However, in situations where the upper bound point estimate approaches show 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.

4.3.1 Characterization of Site and Non-Site Source and Media Data for the Exposure Assessment

The exposure assessment step of the current HHRA utilized the available soil and groundwater chemical characterization data from within the study boundary compiled from the Phase II ESA field program (Dillon, 2022b), the supplemental ESA (Dillon, 2023b), and the additional site investigation work conducted in 2023 (**Appendix C**). In addition, literature based rural ambient air data and typical dietary exposure data for the COPCs were included in the multimedia multi-pathway HHRA approach.

4.3.2 Soil

To evaluate the level of soil exposure for receptors to each potential COPC, it is necessary to first estimate the representative concentration of each COPC in soil. Where data are sufficient in terms of sample size, Health Canada (2021a) recommends use of the 95% upper confidence limit of the arithmetic mean (UCLM95) to assess exposure (i.e., the UCLM95 is applied as the exposure point concentration (EPC)). The UCLM95 is the most common statistic used to represent EPCs in HHRAs. Most regulatory agencies recognize that 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. Accurate and statistically robust estimates of the UCLM95 are important in an HHRA. To derive the UCLM95 values, the U.S. EPA computer program, ProUCL™ Version 5.2 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 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 eight. As the sample size for COPCs within the study boundary soil chemistry dataset is larger than this (i.e., $N > 100$ for lead and strontium in surface soils), it was possible to calculate adequately robust UCLM95 soil concentrations for each COPC.

In calculating the UCLM95 soil concentrations for the various COPCs (if/as necessary), the following tasks/conditions were conducted/applied, all of which tend to bias the UCLM95-based EPCs high:

- For soil samples within the study boundary with corresponding laboratory or field duplicates, the higher concentrations out of the original and duplicate samples were retained;
- For soil samples collected from the same locations, but at different depths, the sample depth most relevant for human exposures was retained (i.e., generally the top 15 cm) as the site is anticipated to remain as it is with no construction occurring on the site;
- For any study boundary soil samples with analytical results for the various COPCs below the laboratory reported detection limit (i.e., <RDL), the <RDL values were assumed to equal the RDL;
- Prior to calculating UCLM95-based EPCs, Dillon reviewed the laboratory certificates of analysis for relevant data, as well as the accompanying laboratory quality assurance reports for these data. This review focused on laboratory performance with respect to the RDLs that were achieved, % surrogate recoveries, lab and field duplicate results and relative per cent difference or absolute difference (when lab duplicates are compared to original sample results), matrix spikes, method blanks, and spiked blanks. No major analytical issues were identified that would affect the use of these data in an HHRA. Thus, the study boundary soil chemistry data were considered to be of adequate quality for use in an HHRA and appropriate for the purposes of EPC calculation;
- As the measured COPC concentrations in study boundary 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 site soil or chemistry datasets. Thus, the EPC calculations for the COPCs included the presence of potential extreme values; and
- As the calculated options for a UCLM95 generated by ProUCL 5.2 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), any warnings generated by the ProUCL 5.2 software, and the magnitude of the calculated UCLM95 options. ProUCL UCLM95 outputs are provided in **Appendix I**.

4.3.3 Groundwater/Surface Water

While site groundwater is non-potable, off-site groundwater wells are present within approximately 100 m of the site. While not confirmed, the cottages along the eastern shore of Lake Enon could also potentially use surface water from the lake for potable purposes such as drinking and cooking. As such, surface water data from Lake Enon (with the exception of the one sample collected at the outflow of the study boundary) and groundwater data collected within the study boundary were considered in selecting groundwater EPCs for the COPCs.

Measured surface water concentrations in Lake Enon for lead and strontium were lower than measured groundwater concentrations from within the study boundary. As such, the data used to determine potential drinking water EPCs for strontium and lead were the site groundwater data.

Groundwater reportedly flows from the site into Lake Enon. Given this, the most representative groundwater samples for drinking water would be from wells that flow towards the residential properties (i.e., MW8, 7.8 m depth), rather than wells that flow into Lake Enon. Concentrations of lead and strontium at MW8 were <0.00050 mg/L and 1.2 mg/L, respectively. When groundwater data across the site are averaged, the mean strontium groundwater concentration is 24.2 mg/L, while the mean lead concentration (assuming the non-detectable concentrations are equal to the detection limit) equals 0.0013 mg/L. The EPCs for lead and strontium in drinking water were conservatively selected to be 0.0013 mg/L and 24.2 mg/L, respectively.

4.3.4 Ambient Outdoor Air

No ambient air monitoring data were available in the vicinity of the site. Thus, a number of other data sources and documents were reviewed in an effort to identify representative background ambient air concentrations for lead and strontium for the Lake Enon area. The resources that were searched included the following:

- NAPS Program data for rural stations that measure metals and metalloids on PM2.5;
- Selected scientific literature if/as necessary;
- CCME science support documents for soil quality guidelines;
- Health Canada drinking water quality guideline science support documents;
- WHO INCHEM documents (including any relevant JECFA documents, EHC monographs and CICADs);
- ATSDR toxicological profiles;
- USEPA integrated science assessments (ISAs) for air contaminants; and
- The USEPA IEUBK Pb model documentation that pertains to model inputs, parameters and assumptions.

Details on the selection of the lead and strontium ambient air concentrations are provided in the following paragraphs.

Lead

A background rural ambient air lead concentration of 0.001 µg/m³ was used in the exposure assessment. This value is reported within USEPA (2023; 2013) as the upper end of a plausible natural background range of ambient air lead concentrations (i.e., 0.02 to 1 ng/m³, or 0.00002 to 0.001 µg/m³). The upper end of this range (0.001 µg/m³) was selected to represent typical ambient air levels of lead in a rural location (Lake Enon is a rural area) with no significant point or mobile sources of lead air emissions. The selected air lead concentration is supported by review of select rural Canadian NAPS program data for lead (for the stations that analyze metals and metalloids on PM2.5). The most recent

accessible data posted online at the NAPS Program webpages are for 2019, which are considered reasonably representative of potential current ambient air lead concentrations (<https://donnees-data.ec.gc.ca/data/air/monitor/national-air-pollution-surveillance-naps-program/Data-Donnees/2019/IntegratedData-DonneesPonctuelles/?lang=en>).

Only two NAPS stations collected data for lead in 2019 that can be considered representative of rural locations in Canada, as follows. Most NAPS stations are sited in urban or industrialized areas with known or potential air emissions of interest to the NAPS Program, and do not represent rural ambient air quality. Only NAPS stations that represent rural air quality are relevant to the HHRA, as the Lake Enon area is very rural.

- Station 62601 in Simcoe, ON: This station is used as an Ontario background air quality station that is located in an area of agricultural land use. The station represents a population density of 500 to <10K people within a 4 km radius of the station; and
- Station 129003 in Yellowknife, NT: This station is located in a residential area of Yellowknife. The station represents a population density of 10K to <50 K within a 4 km radius of the station.

While the Lake Enon area has a much lower population density within a 4 km radius than these two NAPS stations represent, the identified stations are the most rural of the NAPS stations that collect data on metals and metalloids in ambient air PM_{2.5}.

At the Simcoe station in 2019, there were only 6 quantifiable analytical results for lead out of 70 PM_{2.5} samples that were collected. Lead was present in PM_{2.5} below detection limits in most samples, and there were also a number of instances of no lead data being collected due to air monitoring equipment errors or failures. For the 6 PM_{2.5} samples with measurable lead concentrations, the range was 0.0002 to 0.001 µg/m³ with an average concentration of 0.0006 µg/m³. At the Yellowknife station in 2019, there were only 10 quantifiable analytical results for lead out of 66 PM_{2.5} samples. Lead was present in PM_{2.5} below detection limits in most samples, and there were also a number of instances of no lead data being collected due to air monitoring equipment errors or failures. For the 10 PM_{2.5} samples with measurable lead concentrations, the range was 0.0005 to 0.003 µg/m³ with an average concentration of 0.001 µg/m³. Thus, the limited NAPS station data support the use of the 0.001 µg/m³ natural background air lead concentration reported by the USEPA (2013; 2023).

Strontium

For strontium, there are limited ambient air concentration data reported in regulatory documentation and the scientific literature. Thus, for the HHRA exposure assessment, the average air strontium concentration between the Simcoe and Yellowknife NAPS stations (noted above) was used to represent a rural background ambient air strontium concentration (i.e., 0.0017 µg/m³).

At the Simcoe station in 2019, there were 30 quantifiable analytical results for strontium out of 70 PM_{2.5} samples that were collected. Strontium was present in PM_{2.5} below detection limits in most

samples, and there were also a number of instances of no strontium data being collected due to air monitoring equipment errors or failures. For the 30 PM_{2.5} samples with measurable strontium concentrations, the range was 0.00032 to 0.0039 µg/m³ with an average concentration of 0.0014 µg/m³. At the Yellowknife station in 2019, there were 14 quantifiable analytical results for strontium out of 66 PM_{2.5} samples. Strontium was present in PM_{2.5} below detection limits in most samples, and there were also a number of instances of no strontium data being collected due to air monitoring equipment errors or failures. For the 14 PM_{2.5} samples with measurable strontium concentrations, the range was 0.00043 to 0.0074 µg/m³ with an average concentration of 0.002 µg/m³.

Although very little recent reliable data were identified from other resources on rural ambient air concentrations of strontium, the air concentration ranges reported in ATSDR (2004) and HC (2019) generally supports the value of 0.0017 µg/m³ that was used in the exposure assessment.

4.3.5 Dietary Estimated Daily Intakes (EDIs) for Lead and Strontium

A number of data sources and documents were reviewed in an effort to identify representative and current dietary EDIs for lead and strontium. Dietary EDIs are not specific to any particular site or study area and are typically assumed to be the same for any individual living anywhere in Canada, as they reflect potential exposures from common market basket (grocery) food items that are readily available across Canada. Dietary EDIs are important to consider in HHRAs for any COPC that is known to be readily present in the typical Canadian diet, as dietary exposures can be significant and may be considerably higher than COPC exposures incurred at any given contaminated site. The resources that were searched included the following:

- CCME science support documents for soil quality guidelines;
- Health Canada drinking water quality guideline science support documents;
- WHO INCHEM documents (including any relevant JECFA documents, EHC monographs and CICADs);
- ATSDR toxicological profiles;
- U.S. EPA Integrated science assessments (ISAs) for air contaminants;
- The USEPA IEUBK model documentation that pertains to model inputs, parameters and assumptions;
- Health Canada Total Diet Study (TDS) results (data summaries and reports);
- U.S. FDA Total Diet Study (TDS) results (data summaries and reports); and
- Selected scientific literature if/as necessary.

Ultimately, Health Canada TDS data reported for several cities (Vancouver, Winnipeg, Halifax, Toronto, Montreal) across Canada from 2003 to 2007 were used to develop dietary EDIs for lead and strontium, for all assessed receptor age classes. The Health Canada TDS program up to 2007 reported EDIs for chemical contaminants for multiple human receptor age classes (in dose units of µg/kg BW/day), for both males and females, and this information was utilized directly to develop dietary EDIs for lead and strontium for the HHRA. Specifically, the midpoint EDI values from all male and female receptor-specific

EDIs reported for 2003 to 2007 were used to develop the dietary EDIs for lead and strontium that were used in the HHRA. A summary of the lead and strontium dietary EDIs by receptor type is provided in **Table 4-7**.

Table 4-7: Dietary Estimated Daily Intakes (EDIs) for Lead and Strontium

| Receptor Type | Dietary EDI ($\mu\text{g}/\text{kg BW}/\text{day}$) |
|--|---|
| Lead Dietary EDIs ($\mu\text{g}/\text{kg BW}/\text{day}$) | |
| Male and Female Toddler | 0.24 |
| Male and Female Child | 0.17 |
| Male and Female Adolescent | 0.12 |
| Male and Female Adult | 0.115 |
| Strontium Dietary EDIs ($\mu\text{g}/\text{kg BW}/\text{day}$) | |
| Male and Female Toddler | 63.5 |
| Male and Female Child | 43 |
| Male and Female Adolescent | 25.5 |
| Male and Female Adult | 22.5 |

Health Canada's practice of providing receptor-specific EDIs appears to have ceased as of 2007, although Health Canada TDS programs continue to occur in various Canadian cities since 2007. As the Health Canada 2003-2007 TDS results are somewhat dated, more recent information (where identified) from the noted resources was used to validate or groundtruth the estimated EDIs that were determined from the TDS data summaries. Recent drinking water quality guideline supporting scientific documentation is available from Health Canada (2019a, b) for both lead and strontium. Dietary EDI information is presented within both of these documents, which supports the dietary EDIs determined from the 2003 to 2007 TDS data, in that very similar dietary EDI values were noted in Health Canada (2019a,b) relative to those presented in **Table 4-7**. Support for the selected dietary EDI values for lead also comes from review of the most current documentation for the U.S. EPA IEUBK model for lead, which reports similar ranges of dietary EDIs as those that were developed from the Health Canada TDS data (when adjusted for receptor body weights, as the IEUBK model documentation does not report dietary EDIs on a receptor-specific dose or exposure rate basis). In addition, though also somewhat dated, the ATSDR (2004) toxicological profile for strontium also reports similar dietary EDI values to those presented in **Table 4-7**.

4.3.6

Selected Exposure Point Concentrations (EPCs) for Exposure Modelling and Exposure Estimation

The selected EPCs for lead and strontium are provided in **Table 4-8** below, along with a brief description for the selected EPC values. The ProUCL outputs for soil EPC calculation are provided in **Appendix I**. Dietary EDIs used in the HHRA are previously presented in **Table 4-7**.

Table 4-8: Exposure Point Concentrations for COPCs Evaluated in the HHRA

| COPC | EPC Value | Source/Description |
|-----------------------|--------------------------|--|
| <i>Soil</i> | | |
| Lead | 4369 mg/kg | UCLM 95 95% Hall's Bootstrap UCL |
| Strontium | 5917 mg/kg | UCLM 95 95% Chebyshev (Mean, Sd) UCL |
| <i>Drinking Water</i> | | |
| Lead | 1.3 µg/L | Average concentration of groundwater data from site boundary (including wells that flow towards Lake Enon) |
| Strontium | 24,200 µg/L | |
| <i>Air</i> | | |
| Lead | 0.001 µg/m ³ | US EPA, 2013; 2023 (rural air data) |
| Strontium | 0.0017 µg/m ³ | NAPS, 2019 (rural station air data) |

Exposure Equations:

Exposure equations based on those provided within Health Canada (2021a) were used to estimate exposures to lead and strontium within the HHRA. **Appendix I** provides the equations, factors, parameters, and assumptions used in the exposure and risk calculations.

4.4 Toxicity (Hazard) Assessment

Toxicity is the potential for a chemical agent to produce any type of damage, permanent or temporary, to the structure or function of any part of an organism. The toxicity of a chemical depends on the amount that is taken into the organism (referred to as the “dose”) and the duration of exposure (i.e., the length of time the person or other 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 a HHRA, information relating to the dose-response relationship of each COPC is evaluated (usually from laboratory animal studies and epidemiological studies of human exposure in the workplace) to determine the maximum dose to which humans can be continuously exposed that would be associated with no or a very low probability of 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.

4.4.1.1 Toxicity Reference Values

TRVs are typically derived by regulatory agencies based on detailed reviews of toxicological, epidemiological, and other scientific information, professional judgment, and technical oversight by a number of experienced scientists with expertise in the toxicological sciences. They are often derived to be protective of the most sensitive endpoints in individuals (e.g., organ damage, neurological effects, cancer, or reproductive effects), 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 effect level [NOAEL]). Thus, exceedance of a TRV does not necessarily mean that adverse effects will occur; rather, it means that 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. TRVs derived for threshold chemicals that are not believed to be potential carcinogens are typically expressed as dose (e.g., $\mu\text{g}/\text{kg}$ body weight/day); and
- **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/assumed for carcinogens (although, some carcinogenic substances do have a well-defined threshold below which cancer does not appear to be a response to exposure). In theory, any exposure to a non-threshold substance has the potential to cause damage. As such, 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 target cancer risk level of one-in-one hundred thousand to one-in-one million. Essentially, these numbers correspond to 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 or target level of carcinogenic risk is a policy rather than a scientific decision, and is set by regulatory agencies, as opposed to risk assessors. TRVs derived for non-threshold chemicals that are believed to be potential carcinogens are typically expressed as cancer slope factors or cancer potency factors [e.g., $(\mu\text{g}/\text{kg}$ body weight/day) $^{-1}$], or unit risk values for environmental media [e.g., $(\mu\text{g}/\text{m}^3)^{-1}$]. However, TRVs for carcinogens may also be expressed as risk-specific media concentrations or doses that are associated with a particular level of acceptable cancer risk.

Lead, one of the COPCs in the current HHRA, is a unique substance in that it is not believed to have a threshold below which adverse effects do not occur, but the toxicological endpoints for lead are not related to carcinogenicity. No known regulatory agency has developed guidance to assess lead given its unique non-threshold, non-carcinogen status, but in practice, most risk assessment professionals have opted to assess lead as if it were a threshold substance, and then account for its non-threshold status

during risk characterization and when developing risk management recommendations. That was the approach taken herein.

Health Canada (2021a) states that for the assessment of risks posed by chemicals found at federal contaminated sites in Canada, Health Canada TRVs should be employed, when available, for the characterization of potential health risks. While the Lake Enon property is provincially owned, Nova Scotia follows the federal guidance for human health risk assessment, and Health Canada TRVs were preferentially used where available.

TRVs for the identified COPCs are summarized in **Table 4-9**. Health Canada TRVs were selected where available. TRVs are generally developed to be protective of all members of a human population including sensitive life stages (such as the elderly and/or pregnant women) and individuals of compromised health, where data allow.

Table 4-9: Summary of Toxicity Reference Values (TRVs) Used in the Human Health Risk Assessment

| COPC | Exposure Route | Toxicity Reference Value (TRV) | Health Endpoint(s) | Regulatory Agency |
|-----------|---|--|--|---|
| Lead | All (oral; inhalation) | RsD = 0.5 µg/kg BW/day (provisional) | Neuro-developmental toxicity; cognitive function; (from European Food Safety Authority, 2013, based on Lanphear et al., 2005). | Health Canada, 2021b |
| Strontium | Oral (the oral TRV was also used for the inhalation pathway via exposure route extrapolation as no identified regulatory agency has developed a valid inhalation TRV for strontium) | TDI = 1417 µg/kg BW/day | Decreased bone mineralization in young rats (Marie et al., 1985). | Health Canada (Guideline for Canadian Drinking Water Quality), 2019 |

Notes:

TRV=toxicity reference value; BW= bodyweight; TDI = Tolerable Daily Intake; RsD = Risk-Specific Dose.

4.4.2 Bioavailability, Bioaccessibility and Route Extrapolation Considerations

4.4.2.1 Bioavailability

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 that are ingested, inhaled, or come into contact with the skin are actually absorbed into the body's systemic circulation. 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). Relative bioavailability is the most important type in the context of HHRA. It 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).

4.4.2.2

Selection of Oral, Inhalation and Dermal RAFs for the HHRA

This HHRA conservatively assumed an oral RAF of 1.0 for strontium. This is equivalent to assuming that the oral bioavailability of strontium in soil is the same as the oral bioavailability of strontium in the toxicity studies from which the oral TRVs were developed.

Oral toxicity of a soil-borne chemical depends on the fraction absorbed from the gastrointestinal tract into the systemic circulation and that reaches target organs and tissues. In an HHRA, the bioavailability of a chemical describes the absorption and uptake of that chemical into the human circulatory system. In the exposure assessment of an HHRA, external doses of a chemicals are mostly used to estimate the amount of the chemical absorbed into systemic circulation. The portion of chemical absorbed following soil ingestion is assumed to equal the absorption in the toxicity studies used to determine human health benchmarks. This generally overestimates the bioavailability since the toxicity studies commonly use forms of chemicals that are easily absorbed. There can be large differences in metals bioavailability due to a variety of factors including, for example, the form of the metal, chemical interactions, soil weathering and properties of the soil (e.g., pH and organic carbon; Health Canada, 2017).

The fraction (or percent) of a dose that is absorbed is called absolute bioavailability (ABA). Relative bioavailability (RBA) describes the comparative difference in bioavailability between different forms of a substance or different environmental matrices (e.g., food vs. water). Health Canada has published default RBA values for some contaminants including lead; however, using a site-specific RBA value can provide a more realistic estimate of potential exposures and risks. RBA values can be derived using validated *in vitro* bioaccessibility (IVBA) testing. Validated tests for human IVBA analysis are available for lead.

To aid in calculating site-specific oral RBA for lead, five soil samples (plus a duplicate sample) were submitted to the Environmental Sciences Group at Royal Military College (RMC) of Canada for IVBA testing. The IVBA test methods used have been validated and approved by Health Canada. IVBA results are provided in **Appendix G**.

One soil sample (SSS32) submitted for IVBA analysis was collected from AEC 2 (waste rock dump) which had a percent bioaccessibility (%BA) of 70%. In addition, two soil samples (SSS40; SSS36) were collected from AEC 10 (tailings disposal area; %BA of 75% and 98%, with an average of 86.5%); one soil sample (SSS29) was collected from AEC 9 (waste rock dump; but close to AEC 3, the processing area; %BA of 61%); and 1 sample and 1 duplicate soil sample (SSS28; SSS28-duplicate) were collected in the middle of AECs 3, 4 and 7 (processing area, mill plant area and settling ponds; %BA of 72% and 69% with an average of 70.5%) (**Appendix G**). Soil sample locations are shown in **Figure 3, Appendix A**.

The %BA results from the various AECs ranged from 61% to 98%, with the highest bioaccessibility occurring in the samples from the tailings disposal area. The oral bioaccessibility for the human health exposure modelling for lead was conservatively assumed to be an average of the 2 samples collected within the tailings disposal area (average of 86.5%).

To convert the %BA to IVBA, the average %BA for the two tailings disposal area samples was divided by 100, resulting in an IVBA factor of 0.865.

Using the following Health Canada (2017) equation for calculating the relative bioavailability (RBA) lead, the resulting RBA value was 73%. This oral RBA value was used as the RAF_{oral} in the HHRA, where the RAF is the relative absorption factor.

$$\begin{aligned} RBA_{lead} \text{ (or } RAF_{oral}) &= (0.878 \times IVBA) - 0.028 \\ &= (0.878 \times 0.865) - 0.028 = 0.7332 \end{aligned}$$

Further details regarding the IVBA testing of study boundary soil samples for lead are provided in **Appendix G**.

The HHRA assumed that soil/dust inhalation exposures to the COPCs were 100% bioavailable. This is consistent with Health Canada (2021a) guidance.

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

4.4.2.3

Exposure Route Extrapolation

It is necessary to consider exposure route extrapolation when a TRV is not available for the exposure route of interest and no other data (such as toxicokinetics) are available. For example, it is common in HHRAs 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 relative absorption factor (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 the COPCs evaluated in this HHRA, dermal RAFs have been determined by regulatory agencies as follows (**Table 4-10**). This table also summarizes the oral and inhalation RAFs that were assumed for the HHRA.

Table 4-10: Summary of RAFs Used in the HHRA

| COPC | Relative Absorption Factor (RAF) | | |
|-----------|----------------------------------|--------------------|----------------|
| | Oral | Dermal | Inhalation |
| Lead | 0.73 ^a | 0.006 ^b | 1 ^c |
| Strontium | 1 ^e | 0.1 ^d | 1 ^c |

Notes:

- Site-specific; based on soil IVBA testing results.
- Health Canada, 2021b.
- Conservatively assumed that $RAF=1.0$. The HHRA assumed that inhalation exposures to the COPCs were 100%.
- Assumed based on literature review and Health Canada (2021b) and MECP (2011) dermal RAF recommendation for barium, which is chemically similar to strontium.
- Assumed.

It is also noted that for strontium, no reliable inhalation TRVs were identified from the regulatory sources that were reviewed. Only oral TRVs were identified for strontium. Thus, in such situations where an inhalation TRV is not available, it is common to conduct oral to inhalation route extrapolation, where it is assumed that the effects which may occur following oral exposure would also occur following inhalation exposure. However, such extrapolation is only valid if the endpoints for the oral TRV are systemic in nature and not specific to the oral route of exposure only (e.g., effects on the gastrointestinal tract). For the strontium oral TRV, the effects are systemic (decreased bone mineralization); thus, application of the oral TRV for strontium for inhalation exposures was considered appropriate and reasonable.

Risk Characterization

Risk characterization is the final step in a HHRA. It integrates the exposure and hazard (toxicity) assessments to estimate the likelihood of potential 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 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 (typically non-carcinogenic effects), this comparison takes the form of a ratio calculation, and is referred to as the hazard quotient (also sometimes referred to as an exposure ratio (ER) or risk quotient (RQ)). The hazard quotient (HQ) is calculated by dividing the estimated level of total exposure by the TRV, as follows.

$$\text{Hazard Quotient} = \frac{\text{Estimated Exposure } (\mu\text{g/kg body weight/day})}{\text{TRV } (\mu\text{g/kg body weight/day})}$$

For chemicals with TRVs based on non-threshold dose response relationships (such as carcinogens), risk characterization typically involves determining an incremental lifetime cancer risk (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 question². 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 assessed 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 target cancer risk levels (TCRLs) to evaluate the significance of the ILCR. In Nova Scotia, the TCRL is one-in-one hundred thousand (i.e., 1×10^{-5} , or one additional cancer case per hundred thousand people).

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

$$\text{ILCR} = \text{Estimated Exposure } (\mu\text{g/kg body weight/day}) \times \text{CSF } ([\mu\text{g/kg body weight/day}]^{-1})$$

Where:

| | | |
|------|---|----------------------------------|
| ILCR | = | Incremental Lifetime Cancer Risk |
| CSF | = | Cancer Slope Factor |

² 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.

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

In HHRA, it is also common to assess COPCs that have available TRVs which are based on both cancer and non-cancer endpoints, as both a carcinogen and a non-carcinogen, and to determine human health risk estimates for both types of endpoints.

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:

- Comparisons of potential adverse health effects between COPCs and between different exposure scenarios (e.g., different site-specific conditions, different site use options);
- Estimation of 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, or kidney toxicity, or respiratory tract cancers, or respiratory irritation effects); and
- Simplification of HHRA results to provide clear understanding of the results, and an appreciation of their significance.

None of the COPCs in the current HHRA are carcinogenic, thus only HQ values are presented in the HHRA results section (i.e., **Section 4.5.3**) 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 provided in **Section 4.7**.

4.5.1 Interpretation of HQs

Once HQ values have been determined, they are then compared to a target HQ (THQ) 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 THQ would be 1.0 or less, and no adverse health effects would be expected. In this case, the THQ would be 1.0, assuming that there are estimates of exposure from all relevant exposure pathways and sources.

However, in 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 THQ cannot be 1.0 as a number of pathways would not be considered. Rather, a lower THQ would be selected. The reason for this is that non-carcinogenic (threshold chemical) 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, the selection of a THQ of 1.0 may not be appropriate or adequately protective.

In many HHRA's conducted across Canada, a default THQ of 0.2 is typically used when the assessment is only investigating exposure from relatively few selected pathways, sources, and routes. This implies that 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, diet, and consumer products). The default THQ of 0.2 has been endorsed and/or recommended by various regulatory agencies in Canada for many years when limited exposure pathways, sources, and routes are under investigation. This does not mean, however, that a higher THQ value cannot be used for certain substances in a given contaminated site HHRA, so long as the rationale for a higher THQ value can be justified in a technically defensible manner. For example, if all key site and at least some key non-site (background) sources and/or pathways of exposure can be accounted for in a HHRA, or if there is information to demonstrate that certain site-related or non-site-related exposure pathways for a COPC are negligible, then there would be less justification for a THQ of 0.2 and a higher THQ could be applied.

In the current HHRA, all relevant site sources and exposure pathways are considered (i.e., soil contact pathways and groundwater ingestion; other potential site-related exposure pathways are negligible with respect to COPC exposure), and two key non-site (background) sources and pathways of exposure for the COPCs are also considered (i.e., ambient air inhalation and typical dietary exposures (represented by dietary EDIs)). The only source of potential COPC exposure that is not considered in the HHRA is consumer products. Though very few consumer products contain strontium, and the lead content within consumer products has declined considerably over time, a THQ of 0.8 (rather than 1.0 or 0.2), was applied in the HHRA to account for the fact that one category of human exposure was not considered [i.e., 1 of 5 categories = 20% or 0.2; thus, the THQ drops from 1.0 to 0.8].

In general, if non-cancer risk estimates (i.e., HQ values) are less than the THQ in a HHRA, no adverse health effects would be expected to occur given the COPCs, receptors, exposure pathways and scenarios evaluated. As HHRA's typically utilize 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 the THQ, 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 HHRA's, calculated HQ values greater than the THQ value do not necessarily indicate that adverse health effects are likely to occur. Generally, in cases where HQ values are greater than the THQ in an HHRA, assumptions, parameters and data used within the HHRA are closely re-examined prior to concluding whether or not a human health risk exists.

4.5.2 Other Risk Characterization Considerations

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 THQ or TCRL 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 and/or risks can be summed together. This was the case for both COPCs in the HHRA, wherein strontium only has an oral TRV available, and the TRV for lead is based on an internal dose (i.e., blood lead concentrations) where the key endpoints are the same for all potential pathways and routes of exposure.

With respect to potential chemical mixture effects and the potential for toxicological interactions between the COPCs assessed in the HHRA, and between the COPCs and other substances, the following standard HHRA approach was taken. That is, the HHRA evaluates the COPCs individually, and then determines whether or not the exposures or risks for the individual COPCs could reasonably be considered additive, based on the health effects associated with each substance. Other potential types of toxicological interactions are typically not considered in HHRA as the data to assess such interactions is generally not available or is insufficient to quantify the type, extent, and direction of the potential interaction. As the health endpoints associated with the TRVs for both lead and strontium are different (i.e., neurodevelopmental toxicity and impaired cognitive function for lead; decreased bone mineralization for strontium), neither exposures nor risks for these COPCs were considered to be additive.

The fact that lead is a COPC in the HHRA merits some special risk characterization considerations. The unique status of lead as a non-threshold non-carcinogen with a low toxicity reference value results in the need to consider key non-site sources and pathways of lead exposure (as well as the key site-related sources and pathways of exposure) in order to enable reasonably thorough exposure and risk estimation, and exposure pathway attribution, and to also enable the development of reasonable risk management objectives (such as media-specific site-specific target levels or SSTLs) for a given site that reflect total lead exposures and lead exposure pathway attribution for both site and non-site pathways. As such, a multi-media multi-pathway approach was used for lead in this HHRA. When taking this approach for lead, it is also important to take the same approach for other COPCs being assessed (where data are available to enable such an assessment approach), in order to ensure consistency in the HHRA. The same approach was taken for strontium in this HHRA.

Other considerations that can be important when interpreting HQ and ILCR values include considering other potential sources of COPC exposure, and 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.

The results of the risk characterization for the HHRA are provided in the following section.

4.5.3

HHRA Results

Results of the HHRA for the Outdoor Site Visitor Scenario are summarized in **Table 4-11**. The risk characterization used a multimedia and multi-pathway approach in that exposures via site soil and drinking water ingestion, and background air inhalation and food ingestion were included in the generation of exposure and risk estimates. **Appendix I** provides the equations, factors, parameters, and assumptions used in the exposure and risk calculations.

Table 4-11: Summary of Human Health Risk Estimates

| COPC | Receptor | Hazard Quotient (HQ) ^a | Benchmarks of Safety (Target HQ) |
|-----------|----------|-----------------------------------|----------------------------------|
| Lead | Toddler | 3.6 | THQ = 0.8 |
| | Child | 0.79 | |
| | Teen | 0.51 | |
| | Adult | 0.47 | |
| Strontium | Toddler | 0.093 | |
| | Child | 0.061 | |
| | Teen | 0.039 | |
| | Adult | 0.043 | |

Notes:

Risk estimates incorporate UCLM95 site soil concentrations of COPCs and average site groundwater concentrations. The risk estimates are rounded to two significant figures.

Bolded values indicate exceedance of target HQ.

^a. It is standard HHRA practice to sum oral, dermal and inhalation risk estimates when a COPC is associated with the same or similar effects via both the oral and inhalation routes, or a COPC only has an oral TRV available.

As shown in **Table 4-11**, the THQ for lead was exceeded for only the toddler receptor. Risk estimates for lead for the child, teen and adult receptors were all below the THQ.

All the strontium HQs were below the THQ of 0.8 for all receptors.

Given the risk estimates presented in **Table 4-11**, there is a potential risk to toddlers in relation to lead. However, the HQ of 3.6 for the toddler is considered an overestimate. This is largely because it is extremely unlikely that toddlers would be present within the study boundary for any significant duration. The study boundary also has no unique features that would attract or encourage toddlers and their caregivers to be spending time in the lead-impacted portions of the study area. The HHRA conservatively assumed that a toddler would be on the site once per week for 6 months of the year. However, the likelihood of this occurring is considered negligible. The site and study boundary considered in the HHRA is not easy to access for the public and has a number of physical hazards such as unfenced waste rock piles, settling and tailings ponds. The study boundary also contains wetlands and dense forests in some areas. Much of the study boundary was difficult to traverse for Dillon field personnel and this would be even more the case for a toddler. It is highly unlikely that the study boundary would be used for recreational purposes, particularly with young children, when there are

numerous other areas in the vicinity of the site and around the shores of Lake Enon that would be far more suitable for families engaging in recreational activities.

Section 4.6 discusses proposed risk management recommendations for the study boundary.

4.6 Risk Management Recommendations

When risk characterization results of HHRA indicate potentially unacceptable risks for human receptors, site-specific target levels (SSTL) for site media can be generated as part of the risk management plan. SSTLs are typically iterated from the HHRA exposure and risk model and provide the maximum allowable concentration of a COPC in site media, based on the assessed exposure scenario, which will not pose a risk to human receptors. SSTLs are often used to guide remediation or risk management activities, if/as warranted.

As only site study boundary soil is significantly impacted by COPCs (and only by lead), such that a potential risk was identified for human receptors, only a soil SSTL was determined for lead. Strontium did not pose a risk to any human receptor and even maximum measured study boundary soil concentrations of strontium (i.e., 27,000 mg/kg) did not pose a potential human health risk. Thus, strontium does not require a soil SSTL. Soil remediation based on lead will remove many of the higher soil strontium concentrations as well, since elevated lead and strontium in many site soil samples are co-located.

As noted in the preceding section, only the toddler receptor had risk estimates exceeding the THQ of 0.8. Thus, despite the fact that toddler exposure and risk estimates are likely substantial overestimates of actual exposure and risk and are likely not a reasonable basis for risk management or remedial decision-making, the toddler receptor was conservatively used as the basis for soil SSTL derivation. Iteration within the HHRA model indicates that a soil EPC for lead of 430 mg/kg would not result in the THQ being met or exceeded for the toddler. As the EPC is a UCLM95 value (i.e., an upper estimate of central tendency), a reductive iteration exercise was conducted using the study boundary soil data for lead, where upper soil lead concentrations were incrementally removed from the dataset until a UCLM95 value of 430 mg/kg was reached. This exercise showed that removal (via risk management or remediation measures) of soil sample locations containing lead concentrations of >2100 mg/kg resulted in a UCLM95 soil lead concentration of 430 mg/kg being achieved. Thus, the recommended soil SSTL for lead is 2100 mg/kg. This SSTL value is protective of all human receptors that were evaluated in the HHRA.

The method or remedy by which the soil SSTL for lead will be achieved will be provided under separate cover from this HHRA report. However, it is anticipated that some combination of excavation, capping, and fencing of certain areas would likely occur.

Some other risk management recommendations for the site and HHRA study boundary are as follows:

- Some miscellaneous drums and debris were noted by AEC 1 and south of AEC 12. Some of the drums were partially buried and seemed empty. While not a chemical exposure issue, these drums and debris could pose a physical risk to someone walking in the area. It would be prudent to remove the visible debris and drums;
- Preventing the potential flow of water from the on-site culvert into Lake Enon;
- Prevent the movement of contaminated sediments from the tailing ponds / water courses into Lake Enon;
- Develop site-specific health and safety plan for site remediation workers, as required;
- If remedial activities result in freshly exposed waste rock / tailings, these areas should be evaluated for sulfide bearing materials for acid rock drainage potential as warranted. This can be done using a field screen;
- The ingestion of country foods (including fish from Lake Enon) was excluded as an exposure pathway in the HHRA; however, given the additional information collected in 2024 showing the wide-spread contamination of lead in Lake Enon sediments, and measured concentrations of lead in site bivalves, it would be prudent to collect some fish fillet samples (for species potentially consumed by local residents), and analyze the fillet samples for lead and other metals; and
- Similarly, it was assumed that no hunting occurred on the site. If this changes in the future, it would be prudent to conduct game mammal and/or bird edible tissue sampling for species that are potentially harvested and consumed.

4.7

HHRA Uncertainty Analysis and Conservative Assumptions

In any risk assessment, the findings are based on available data from the specific study area or site, and the scientific literature, in conjunction with a number of assumptions. Reasonable efforts are made to ensure the assumptions and data adequately represent conditions at a given site. However, data are often limited, resulting in uncertainty and/or reduced accuracy or confidence in the assessment. Where uncertainty exists, assumptions are made, and data are selected so as to err on the conservative side. The major sources of uncertainty, limitations and conservatism associated with the HHRA are presented below. Overall, given the tendency for the conservative assumptions used in the HHRA to overestimate COPC exposures, it is considered likely that the HHRA has substantially overestimated potential human health risks.

- The exposure scenarios evaluated in the HHRA are considered the most likely (and most conservative) scenarios given the current use and anticipated future use of the site. However, if anticipated future site use conditions change appreciably, then the need to evaluate other exposure scenarios in the HHRA may need to be considered. No HHRA evaluates all possible scenarios that could relate to a given site at some point in the future, and it is not uncommon for new or revised HHRAs to sometimes be necessary if future site use conditions end up deviating considerably from what was anticipated;

- While the site has been reasonably well characterized in terms of distribution of soil metal concentrations, there remains some sampling and spatial coverage uncertainty as sample locations are generally 100 m or more apart from each other. However, further soil sampling (if it occurred) is unlikely to make a significant difference to HHRA outcomes and conclusions;
- Given the natural enrichment of some metals in site soil, there is some uncertainty regarding the distinction “line” between impacted soils within the study boundary and local background areas. However, there is also reasonable confidence that reasonable background concentrations of metals in soil have been identified, and that study boundary assessment activities have discerned impacted soils within the study boundary from un-impacted soils influenced by natural geologic enrichment;
- While no potable groundwater wells are located within the study boundary, it was conservatively assumed that site visitors would consume either well water or lake water that would be represented by the metals levels in study boundary groundwater;
- No local ambient air data for lead and strontium for the Lake Enon area were available, so data for rural NAPS stations in other parts of Canada were used, along with other sources of suggested rural ambient air concentrations. Although there is uncertainty in the representativeness of these data to the Lake Enon area, the selected ambient air concentrations of lead and strontium were generally supported by literature sources that report rural air data for these parameters;
- 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 of 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 lead), the weeks/year term is also typically not considered, leaving only the days/week term to represent exposure frequency and duration. These are inherently highly conservative assumptions, as there is considerable uncertainty surrounding how soil contact exposure actually occurs, and whether or not it is truly event-driven, or more related to time spent on a 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 (e.g., residential), 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 the study boundary and the outdoor site visitor exposure scenario), 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 that is being assessed. Thus, for low access scenarios, 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 none. 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;

- While the HHRA utilized current Health Canada and CCME soil ingestion rates for human receptors, it should be recognized that these rates are no longer supported by the scientific literature and greatly overestimate COPC exposure via the soil ingestion pathway (e.g., Richardson, 2013). Health Canada is yet to update their recommended soil ingestion rates in light of recent literature that supports lower rates for human receptors;
- 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, for soil samples with field and/or laboratory duplicate samples, the higher concentrations of the identified COPC between the original and duplicate soil samples were retained for assessment;
- The HHRA assumed that 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 applies to total suspended particulate (TSP). For airborne dusts of crustal origin, only a small proportion of TSP contains particles small enough to be in an inhalable or respirable range.
- The outdoor dust level was based on Health Canada (2021a) recommended dust levels that represent unpaved roads with vehicular traffic. While there is an unpaved road on the site, it is very infrequently used and as such, the selected dust level likely overestimates actual dust levels that could become airborne and potentially inhaled;
- 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 (or other media) contaminant concentrations on a site. This was the case in the current HHRA, where UCLM95 soil concentrations of the identified COPCs were evaluated as the EPCs. The use of these upper estimates of study boundary soil concentrations as EPCs represents a highly conservative assumption in the HHRA. In reality, elevated media concentrations of lead and strontium are not distributed uniformly across the study boundary but rather, are localized to certain areas within the study boundary. No one accessing the site would ever be exposed to the upper bound estimates of soil COPC concentrations for the entire duration of their time spent on the property;
- The exposure modelling in the HHRA assumed that there is no attenuation, transformation, degradation or loss of lead or strontium in soils within the study boundary;
- 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 lead and strontium in study boundary soils would be limited during heavy rain or high wind events, as well as on days of high heat and humidity, and on very cold days with or without snow or ice cover. These meteorological factors would reduce the receptors' time outdoors on the subject property,

thereby reducing potential exposures. Not accounting for such factors likely overestimates potential exposures and risks;

- There is the potential for some off-site COPC exposures to human receptors via surface water and sediments in Lake Enon as there are cottages (or residential zoned areas) on the lake and some residential properties along Loch Lomond Road. Surface water samples collected to date from Lake Enon did not result in the identification of any COPCs; **Section 4.2.4.2**), but some sediment samples collected from Lake Enon exceeded the CCME residential soil guideline for lead (140 mg/kg). There were limited sediment samples collected in the nearshore area where individuals could be exposed via wading/playing in the water. The sediment ingestion pathway was excluded for assessment in the current HHRA as it was outside the scope of this study. There is uncertainty regarding near-shore sediment concentrations of COPCs. The collection of nearshore sediment data could reduce this uncertainty;
- On-site soils that exceeded the lead SSTL for human health are present at the property boundary (particularly to the north east and south west of the site). While there is potential that lead impacts in soil extend off-site, the evaluation of soil in these off-site areas was outside the scope of this HHRA; and
- The ingestion of country foods such as fish in Lake Enon and wildlife within the study boundary was not considered a relevant exposure pathway as there was no indication that country foods are harvested / gathered in the area. If this is and / or becomes a relevant exposure pathway in the future, these exposure pathways should be assessed.

4.8 HHRA Conclusions

The overall conclusions from the HHRA are reiterated below.

- The THQ for lead was exceeded for only the toddler receptor (HQ=3.6). Risk estimates for lead for the child, teen and adult receptors were all below the THQ;
- All the strontium HQs were below the THQ for all receptors;
- While there is a potential risk to toddlers in relation to lead, the HQ of 3.6 for the toddler is considered an overestimate. This is largely because it is extremely unlikely that toddlers would be present within the study boundary for any significant duration. The study boundary also has no unique features that would attract or encourage toddlers and their caregivers to be spending time in the lead-impacted portions of the study area. The HHRA conservatively assumed that a toddler would be on the site once per week for 6 months of the year. However, the likelihood of this occurring is considered negligible. The site and study boundary considered in the HHRA is not easy to access for the public and has a number of physical hazards such as unfenced waste rock piles, settling and tailings ponds. The study boundary also contains wetlands and dense forests in some areas. Much of the study boundary was difficult to traverse for Dillon field personnel and this would be even more the case for a toddler. It is highly unlikely that the study boundary would be used for recreational purposes, particularly with young children, when there are numerous other areas in the vicinity of the site and around the shores of Lake Enon that would be far more suitable for families engaging in recreational activities;

- **Section 4.6** discusses proposed risk management recommendations for the study boundary. This included the development of a soil SSTL for lead of 2100 mg/kg, which is protective of all assessed human receptors, including the toddler; and
- **Section 4.7** describes various uncertainties and conservative assumptions applied in the HHRA that are collectively believed to result in substantial overestimates of exposure and risk for all human receptors assessed in the HHRA.

5.0

Ecological Risk Assessment

5.1

Ecological Risk Assessment Framework

An ERA evaluates potential effects on non-human (ecological) organisms, populations, or communities in response to human-induced stressors. The framework and methods used for the ERA are described in this section and its associated subsections.

5.1.1

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 FCSAP ERA guidance (FCSAP, 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 environmental media at the Lake Enon Former Mill site warrant risk management for the protection of ecological receptors that may occur on or utilize the properties. The assessment goal for an ERA relates to the management goal. In the current ERA, the assessment goal was to determine if there are potential ecological risks associated with current conditions at the site, and if so, determine what (if any) management or remedial action is needed to reduce environmental risks and liabilities.

The scope of this ERA is limited to the terrestrial and aquatic media at the site. The scope of the ERA is also limited by the chemistry data that are currently available for the on-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 site, nor does it consider the adjacent freshwater water bodies that could potentially be influenced by the current and historical activities at the site, with the exception of Lake Enon. Contamination within Lake Enon was evaluated for freshwater aquatic life receptors. The evaluation of off-site water bodies other than Lake Enon and / or the evaluation of any other land based properties adjacent to the site, was outside the scope of the ERA as per direction of Build NS. The framework and associated methodologies for each step of the ERA is provided below.

The ERA for the site was conducted in accordance with the NS CSRs and associated Ministerial Protocols, where relevant. In addition, the current ERA considered widely accepted ecological risk assessment frameworks, methodologies and guidance published and endorsed by Environment Canada (e.g., CCME, 2020; the FCSAP ERA Guidance Document [2012a] and associated modules, and the US EPA [i.e., US EPA, 1998]). Where necessary, consideration was also given to ERA guidance from other agencies and jurisdictions in Canada and internationally.

ERAs can involve various levels of detail, complexity, and level of effort. The initial ERA framework developed by Environment Canada (CCME, 1996) reflected 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 that 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., CCME, 2020; FCSAP, 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 that 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 that 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 that are selected for assessment, as well as spatial and temporal factors or boundaries that are 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 investigation. 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 an ERA are based on CCME (2020) and FCSAP (2012a) guidance as illustrated on **Error! Reference source not found.**, below.

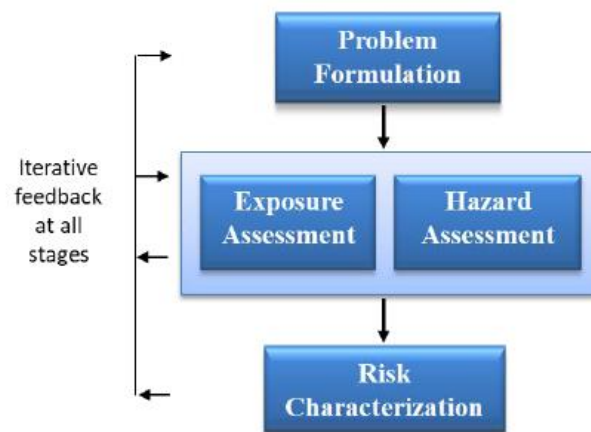


Figure 5-1: Ecological Risk Assessment Steps (from CCME, 2020; FCSAP, 2012a)

Each step of an ERA is briefly described in the following sections.

Problem Formulation: The problem formulation step of an ERA is analogous to that described above for the HHRA and acts as an information-gathering and interpretation stage, which serves to plan and focus the approach of the ERA on the most critical aspects of the site being evaluated.

There are several elements or tasks that typically comprise the problem formulation step 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;
- Selection of study boundary and reference areas;
- Identify receptors of concern (ROC);
- Identify assessment and measurement endpoints;
- Develop lines of evidence (LOE);
- Identify exposure pathways;
- Identify COPCs; and
- Develop a CSM for the ERA.

The outcomes of these tasks form the basis of the approach taken in the ERA. The outcomes of the various problem formulation tasks can also help identify key uncertainties and data gaps that may limit aspects of the ERA, and/or lead to sampling and analytical plan development 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. The outcomes of the problem formulation step for the Lake Enon ERA are provided in **Section 5.2**.

Exposure Assessment: The exposure assessment step of an ERA involves estimating the amount of each COPC 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). Outcomes of the exposure assessment are provided in **Section 5.3**.

Effects Assessment: Also called hazard or toxicity assessment in some jurisdictions, this step identifies toxicity reference values (TRVs) or other types of toxicity benchmarks for each receptor or receptor group evaluated, for each COPC. Outcomes of the effects assessment step are provided in **Section 5.4**.

Risk Characterization and Uncertainty Assessment: Risk characterization typically involves the evaluation and interpretation of 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, other additional site characterization activities, further or supplementary ERA, and/or risk management recommendations or corrective remedial action. This step also involves an uncertainty assessment which provides details of major uncertainties in ERA. Outcomes of the risk characterization and uncertainty assessment step are provided in **Section 5.5**.

5.2 Problem Formulation

As discussed in **Section 5.2**, the problem formulation step is an important information gathering and interpretation stage, which serves to plan and focus the ERA on the most critical aspects of the site being evaluated. An initial ecological health problem formulation was completed for the ERA by Dillon (2023a) in May 2023, which recommended additional sampling and the inclusion of additional site media. The problem formulation herein incorporates the other site media, and the 2023 supplemental sampling and ancillary studies conducted on the site, and relevant data collected in 2024 to confirm and/or update the selected receptors, pathways, and contaminants of concern (COC) for the ERA. Outcomes of the problem formulation are provided in the following sections.

5.2.1 Site Habitat Characterization

Site characterization information has been previously presented in **Section 2.0** and **Sections 3.3** and **3.4** of this report. Additional site information specifically relevant for the ERA are provided below. Photos of the site and the surrounding ecological habitat can be found in **Appendix B** and in **Section 3.2** and **Section 3.3** of **Appendix D**.

While the former mill property is approximately 141 ha, the study boundary focuses on a total area of approximately 51 ha within the former mill site. As described in the Phase II ESA (Dillon, 2022b), the study boundary is generally covered with rock and limited soil. The study boundary is well vegetated with trees and shrubs to the west of AECs 1, 2 and 11 and also to the east of Loch Lomond Road (with the exception of the waste rock/dump [AEC 5]); however, the central areas of the study boundary are less vegetated. Much of the areas within/in the vicinity of AECs 1, 2, 10 and 11 are wetlands and the western areas of the study boundary are densely forested which would limit the use of these areas of the site by some wildlife (see **Figure 4, Appendix D**). Information on potential terrestrial habitat is provided in **Section 3.3.2, Appendix D**.

As noted previously there are two settling ponds and a tailings pond within the study boundary. There is a steep slope along portions of the tailings pond which would potentially limit access by some wildlife.

The settling/tailings ponds were artificially constructed for the mine and were not originally aquatic habitat. Currently, these areas provide some habitat for small fish. During the habitat study, hundreds of minnows were observed in the tailings pond (AEC 6) and small fish (approximately 3 inches long) and smaller minnows in the settling ponds in AEC 7.

Four watercourses and three wetlands were identified during the site visit by Dillon field staff in July 2023. Two of the watercourses were confirmed to have fish, one was considered to provide seasonally accessible fish habitat and the other was considered unlikely to provide fish habitat. In general, site drainage is towards Lake Enon. Additional information can be found in the aquatic habitat assessment report (**Section 3.2, Appendix D**).

Lake Enon is identified as a habitat of potential ecological concern since it is within 200 m of the site (i.e., it borders the site immediately north; Atlantic PIRI, 2012) and a drainpipe connects the settling ponds to the lake. In addition to the settling ponds, the site contains large wet areas with widespread cattail growth and water-logged soil.

5.2.2 Regulatory Context

With respect to the regulatory context for the ERA, the site is located on land owned by the NSDNRR, and as such the NS PSS were preferentially applied. The PSS are benchmarks developed by source agencies and departments outside of Nova Scotia such as the CCME, AEP and MECP. The original guideline sources were reviewed to ensure the NS PSS guideline referenced was the most recent applicable guideline from the original source. Where no NS PSS was identified, other provincial or federal regulatory benchmarks were used, where available. In the absence of Canadian guidelines, the US EPA EcoSSL (Ecological Soil Screening Levels) were considered.

For a number of regulatory jurisdictions, ecological soil quality benchmarks have been derived for specific receptor types such as terrestrial vegetation, soil invertebrates, mammals and birds. In these cases, the lowest benchmark provided was selected for COPC identification purposes.

While the study boundary contains areas with dense forests and wetlands, the areas most impacted within the study boundary site provide limited habitat for ecological receptors as it mostly consists of deforested land and gravel areas (Dillon, 2022b). Based on this, the commercial zoning of the site, and the reasonably anticipated future site use, the most appropriate land use for selecting applicable guidelines for the ERA was determined to be commercial. As such, the COPC screening was conducted using commercial guidelines for ecological health.

Soil within the study boundary ranges from fine to coarse-grained. As such, guidelines for the lower of the guidelines for coarse or fine-grained soils were preferentially sourced from all jurisdictions, where applicable.

5.2.3 Selection of Site Boundaries and Reference Areas

The spatial boundary for the current land-based ERA is the northern, southern, and western property boundary and the study area boundary along the southeastern portion of the property (i.e., approximately 51 ha; see **Figure 2, Appendix A**). Off-site impacts to Lake Enon that could potentially affect aquatic life were also evaluated in the ERA.

Available study boundary and background data used in the ERA have been previously discussed in **Section 2.4**. A summary of the site and background data are provided in **Appendix E**.

5.2.4 Species at Risk (SAR) Assessment

A SAR and terrestrial habitat assessment was completed under the current scope as an ancillary study, and results are described in **Section 3.4** with the full SAR report available in **Appendix D**. The results of the SAR assessment indicated the potential presence of two SAR on the site, as follows:

- Olive-sided flycatcher (*Contopus cooperi*); and
- Snapping turtle (*Chelydra serpentina*).

These SAR were further examined by considering their potential for exposures to site-related COPCs which is influenced by several factors, including the size of the study boundary, availability of food resources within the study boundary, the surrounding habitats and food resources, the home range of the species, and species diet. **Table 5-1** summarizes the potential for site exposures for the SAR evaluated herein. Based on these considerations, the olive-sided flycatcher was not considered to be of concern given its diet (flying insects), foraging habits, and home range size and was not assessed further. The site was considered to offer potential breeding and wintering habitat and food resources for the snapping turtle at AECs 6 and 8; therefore, it was carried forward for further assessment. Potential risks to the snapping turtle are qualitatively discussed in **Section 5.6.5**.

Table 5-1: Discussion of the Exposure Potential of COPCs for Species at Risk within the Study Boundary

| Species | Life History Comments | Available Suitable Habitat within the Site | Potential for Exposure of Site COPCs |
|--|---|---|--|
| Olive-sided flycatcher (<i>Contopus cooperi</i>) | Olive-sided flycatchers are migratory birds. They nest above ground (generally more than 5 feet above ground; CornellLab, 2023) in open, forested areas, often with many conspicuous perches (i.e., tall trees or snags alongside open areas) (ECCC, 2016). | Potential suitable habitat in forested areas (AEC 1, 4, 10 and 12). | Limited potential for exposure given diet of flung insects, foraging many meters above ground, preferred breeding and foraging habitats, and home range size. As such, potential for risk to the olive-sided flycatcher was considered to be low and they were not carried forward for further assessment. |

| Species | Life History Comments | Available Suitable Habitat within the Site | Potential for Exposure of Site COPCs |
|--|---|---|---|
| Olive-sided flycatcher (<i>Contopus cooperi</i>) | <p>Diet is comprised almost entirely of flying insects and often forage for insects near or above the canopy level of surrounding forests (ECCC, 2016).</p> <p>The olive-sided flycatcher is highly territorial with large between 10 and 20 ha (ECCC, 2016).</p> <p>Breeding occurs primarily in forests (ECCC, 2016). As such, breeding would not likely occur in the more contaminated areas of the study boundary.</p> | | |
| Snapping turtle (<i>Chelydra serpentina</i>) | <p>Suitable habitat for the snapping turtle includes slow-moving water with a soft mud bottom and dense aquatic vegetation (ECCC, 2020a). Overwintering sites require water that is shallow enough for the turtle to reach the surface to breathe, but deep enough that it will not freeze to the bottom and mud deep enough for the turtle to bury itself (Meeks and Ultsch, 1990; ECCC, 2020a).</p> <p>Eggs are generally laid on sand or gravel banks near the water, in locations where vegetation is absent or sparse (ECCC, 2020a).</p> <p>Diet includes a variety of vertebrates, invertebrates and plants (NatureServe Explorer, 2023).</p> | <p>Potential suitable wintering/nesting habitat within settling ponds and Lake Enon (AECs 6 and 8). The water within AEC 7 was considered too shallow to offer wintering/nesting habitat.</p> <p>The areas within the study boundary most likely provide a movement corridor for these turtles; however, AECs 6 and 8 could provide nesting or overwintering habitat within the study boundary.</p> | <p>Snapping turtles are omnivorous and opportunistic feeders. Food items can include vascular plants, mollusks, arthropods, fish, and amphibians; however, they generally consume more vegetation than animal matter (Ernst and Lovich, 2009; ECCC, 2020a). Based on their diet and the potential for nesting and overwintering habitat in AECs 6 and 8, snapping turtles, if present within the study boundary, could be exposed to site-related contamination. As such, the snapping turtle was carried forward for further assessment and their potential risk will be qualitatively discussed SAR risk characterization section (Section 5.6.5).</p> |

| Species | Life History Comments | Available Suitable Habitat within the Site | Potential for Exposure of Site COPCs |
|---|--|--|--------------------------------------|
| Snapping turtle (<i>Chelydra serpentina</i>) | Home ranges for male snapping turtles in Ontario were reported to range from a few hectares to a few dozen hectares (ECCC, 2020a). | | |

5.2.5 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 COCs (FCSAP, 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 investigated. 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 of interest, but also considers receptors that may utilize the site for foraging, breeding, nesting, resting and other aspects of their life history.

For the purposes of an ERA, it is neither practical nor necessary to assess each and every species that may potentially occupy or utilize 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 study area characteristics (e.g., surface coverings, habitat types of present, observed species) as determined from environmental site investigations;
- Representation from the various trophic levels, habitats, and feeding guilds that are appropriate for the study area;
- 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, or body weights);
- Habitat quality, suitability, and preferences for receptors that may occur on or utilize 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 within 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 for the study area;
- 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 (e.g., commercial importance of a species, level of value to humans; recognized as 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 on a given site;
- A site visit to visually confirm habitat types, exposure pathways, and the potential for certain wildlife species to occur on or utilize the site; and
- Professional judgement.

The key outcome of the ROC identification step is the consideration of relevant 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 tabular or matrix format. Ideally, the ROCs selected for an ERA will be those that occur within the study boundary (or would be expected to utilize 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 ROCs meet 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.

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 and soil and sediment invertebrates and pelagic aquatic life (e.g., fish, invertebrates), the level of biological organization that is evaluated in an ERA is generally community level (FCSAP, 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 level³ 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 (FCSAP, 2012a; Suter et al., 2000).

³ 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.

Thus, for most ROCs that are 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, or 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 of individuals (such as breeding pairs) that may occur on or utilize a site. Rather, the ecological protection goal is focused on maintenance of local populations of the ROC (or its surrogate), or maintenance of community ecological structure and function.

The selection of ROC considers receptor diet, receptor characteristics, site characteristics (e.g., available habitat), in addition to the size of the site. While the total area of the property is approximately 141 ha, the land based study boundary is approximately 51 ha, and some of the areas within the central areas of the study boundary have limited habitat for some ecological receptors. The study boundary is shown on **Figure 2, Appendix A**. Given that soil contamination on the site has not been fully delineated to the north and south of the study boundary, there is some uncertainty in how far the contamination extends into off-site areas that may have suitable habitat for ecological receptors. The evaluation of off-site soil contamination was not within the scope of this assessment. However, an evaluation of off-site contamination in Lake Enon and potential risks to freshwater aquatic life were considered.

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

- Previous environmental reports (e.g., Dillon 2022a, 2022b, and 2023b as well as historical reports);
- FCSAP (2012b) Ecological Risk Assessment Guidance Module C: Standardization of Wildlife Receptor Characteristics. This document suggests a number of candidate receptors for Canadian federally owned sites;
- Specific guidance related to spatial and habitat quality considerations for ERA (i.e., American Society for Testing and Materials [ASTM], 2014; 2016); and
- Information gathered during fieldwork on the site and obtained through on-line searches regarding wildlife species that would be expected to be within the study boundary.

The ecological receptors and receptor groups that were considered for selection as receptors for the ERA, along with rationale for their inclusion/exclusion are provided in **Table 5-2**. Among the rationale for inclusion or exclusion of ecological receptors is the presence or absence of operable exposure pathways by which receptors could come into contact with COPCs in soil. 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 provided in **Table 5-2**, where relevant. As noted in FCSAP (2012a), 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 5-2: Ecological Receptors Selected for or Excluded from Evaluation in the ERA

| Receptor Groups | Include/ Exclude | Surrogate Species for Receptor Group (if included) | Rationale for Inclusion/Exclusion |
|----------------------------------|-------------------------|---|---|
| Terrestrial Vegetation | Include | Not applicable; assessed at the community level | Terrestrial vegetation has been observed on the site during fieldwork conducted in 2022 (see Appendix B). While there is less vegetation in the central portion of the study boundary, vegetation is plentiful to the west and east of the study boundary. Some eastern areas of the study boundary appear to have been logged some time ago. COPCs in soil may enter vegetation via root uptake and potentially via the pathway of volatilization from soil and subsequent foliar uptake of vapour phase COPCs. Terrestrial vegetation on the site may be consumed by herbivorous wildlife. Terrestrial vegetation is not mobile, and most species have limited strategies to avoid root contact with contaminated soil. Thus, these organisms have a high exposure potential to COPCs in soil and were selected as ROC. |
| Soil Invertebrates | Include | Not applicable; assessed at the community level | Soil invertebrates would be expected to inhabit the site based on physical characteristics. Soil invertebrates (such as earthworms and arthropods) have a key role in soil development and nutrient cycling and are a major food item for many wildlife species. Soil invertebrates are in constant direct contact with soil, are largely immobile, and some species directly consume soil. Thus, these organisms have a high exposure potential to COPCs in soil. |
| Herbivorous Small Mammals | Include | Meadow vole (<i>Microtus pennsylvanicus</i>) Snowshoe hare (<i>Lepus americanus</i>) | Small herbivorous mammals such as the snowshoe hare and woodchucks are some of the herbivorous mammals that are found in Nova Scotia and could possibly be found on the site (Government of Nova Scotia, 2021). Herbivorous small mammals have a high potential to be exposed to COPCs in soil due to burrowing behaviour, ground-foraging of vegetation and seeds, and generally small home ranges. The site is large relative to the home and foraging range sizes of small mammal herbivores (e.g., meadow vole - 0.0069 ha to 0.348 ha; snowshoe hare - 1.6 ha; FCSAP, 2012b). Small herbivorous mammals could be exposed to |

| Receptor Groups | Include/ Exclude | Surrogate Species for Receptor Group (if included) | Rationale for Inclusion/Exclusion |
|---|------------------|--|--|
| Herbivorous Small Mammals | | | <p>COPCs via the consumption of vegetation and incidental soil ingestion. As such, this receptor group was selected as a ROC.</p> <p>The meadow vole and the snowshoe hared were selected as surrogate species for this receptor group as they occur in Nova Scotia and would likely have relatively high exposures due to their small home ranges.</p> |
| Insectivorous or Invertevorous Small Mammals | Include | Masked shrew (<i>Sorex cinereus</i>) | <p>Insectivorous/invertevorous small mammals such as the masked shrew, star-nosed mole and deer mouse are some of the insectivorous/invertevorous mammals that are found in Nova Scotia and could possibly be in the area (Government of Nova Scotia, 2021; NatureServe Explorer, 2023).</p> <p>Insectivorous/invertevorous small mammals have a high potential to be exposed to COCs in soil due to their burrowing behaviour, consumption of soil dwelling organisms (soil invertebrates) and generally small home ranges. The site is large relative to the home and foraging range sizes of small mammal insectivores/invertevores. While some of the habitat that makes up the AECs and immediate surrounding areas would not be considered suitable (e.g., extremely wet areas, piles of waste rock, concrete pad), there are areas that would be suitable habitat for insectivorous small mammals and the site could provide food resources these receptors.</p> <p>This receptor group was included as a ROC and the masked shrew was selected as a surrogate for this receptor group.</p> |
| Herbivorous Birds | Include | Ruffed Grouse (<i>Bonasa umbellus</i>) | <p>Herbivorous birds such as the ruffed grouse and spruce grouse have been reported to occur in Nova Scotia (Nova Scotia Bird Society, 2023). While herbivorous birds tend to have moderately large home/foraging ranges, some areas of the site could contain habitat and food resources for herbivorous birds. Thus, it is possible that herbivorous birds occur/forage on the site where they would be exposed to COPCs via the consumption of vegetation and incidental soil ingestion.</p> |

| Receptor Groups | Include/ Exclude | Surrogate Species for Receptor Group (if included) | Rationale for Inclusion/Exclusion |
|------------------------------------|------------------|--|---|
| Herbivorous Birds | | | The ruffed grouse was selected as a surrogate for this receptor group given that it occurs in Nova Scotia (NatureServe, 2023). |
| Insectivorous/ Invertevorous Birds | Include | Dark-eyed Junco (<i>Junco hyemalis</i>) | <p>Insectivorous/invertevorous birds have a high potential to be exposed to COPCs in contaminated soils due to their typical ground foraging behaviour in relation to the consumption of soil dwelling organisms (soil invertebrates) and generally small home/foraging ranges. The site is large relative to the home and foraging range sizes of most small bird insectivores/invertevores, however there is limited habitat and food resources within the study boundary. Although there is limited habitat and food resources for these receptors, it is likely they could forage in some parts of the site; therefore, insectivorous/invertevorous birds were selected for assessment.</p> <p>The dark-eyed junco was selected as a surrogate for this receptor group to assess COPC exposures to insectivorous bird species. The junco has been reported to be a year round resident of Nova Scotia and is physically smaller in size (Connell Lab, 2023). The junco is a commonly assessed avian receptor in ERAs due to its year round residency, foraging behaviours which tend to increase potential exposure to COPCs in soil and food items, its small body weight, and relatively small home/foraging range.</p> |
| Carnivorous Small Mammals | Include | Ermine (<i>Mustela erminea</i>) | Carnivorous small mammals, such as the red fox and the ermine (or short tailed weasel), could potentially occur on site and be exposed to COCs via the ingestion of prey items that occur or forage on the site, and via incidental soil ingestion from the site. Prey items on the site would consist primarily of herbivorous small mammals and birds. The study boundary area is larger than some home ranges of carnivorous species (e.g., the ermine home range in NS is reported to be approximately 12 ha to 16 ha Natural Resources and Renewables, 2023). As such, carnivorous small mammals were assumed to occur/forage on the site, where they could be exposed to COPCs via the consumption of small mammals and soil (incidentally) and were selected as receptors. |

| Receptor Groups | Include/ Exclude | Surrogate Species for Receptor Group (if included) | Rationale for Inclusion/Exclusion |
|--|------------------|---|---|
| Carnivorous Small Mammals | | | The ermine was selected as a surrogate species for this receptor group as it has been reported to occur in Nova Scotia (Natural Resources and Renewables, 2023). |
| Freshwater Pelagic and Benthic Aquatic Life | Include | Not applicable; assessed at the community level | <p>Small fish were observed in settling/tailings ponds in AECs 6 and 7. In addition, of the four watercourses within the study boundary, two were confirmed to have fish and one was considered to provide seasonally accessible fish habitat.</p> <p>Lake Enon is adjacent to the site and site drainage is generally towards Lake Enon. What appeared to potentially be tailings were noted in portion of the lake adjacent to the study boundary. It is assumed pelagic and benthic freshwater life could be present in Lake Enon and could be exposed to contaminants that originated from the site. Therefore, freshwater aquatic life was carried forward as a receptor.</p> |
| Amphibians | Include | Not applicable; assessed at the community level | Amphibians are commonly regarded as being 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. Parts of the site are more marsh-like and could potentially offer habitat to amphibian species. Amphibians can also be exposed via potentially impacted sediments or surface waters in Lake Enon. Amphibians were therefore carried forward as a receptor. |
| Species at Risk (or other rare, threatened or endangered species) | Include | Snapping Turtle (<i>Chelydra serpentina</i>; qualitatively discussed in Section 5.6.5) | A SAR assessment was conducted for this study (Section 3.4.2) and based on the SAR outcomes, and in consideration of exposure potential to COCs, it was determined that the snapping turtle could potentially be exposed to site COCs if in the areas of the site that have suitable habitat (i.e., AECs 6 and 8). There are limited toxicity data available for reptiles. Potential exposures to snapping turtle was qualitatively discussed in Section 5.6.5. For any other SAR that that can be reasonably expected to occur within the study boundary, surrogate species for the selected receptor types or groups would be representative and protective of potential SAR that are members of such receptor types or groups. |

| Receptor Groups | Include/ Exclude | Surrogate Species for Receptor Group (if included) | Rationale for Inclusion/Exclusion |
|---------------------|--|--|---|
| Reptiles | Include Snapping Turtle (<i>Chelydra serpentina</i> only) | Not applicable; qualitatively discussed | <p>Terrestrial reptile species could potentially be on the site. Reptiles are commonly regarded as being 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. However, 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. As such, reptiles in general were not included for assessment.</p> <p>However, as the snapping turtle was identified as a potential SAR, potential exposures to this receptor were qualitatively evaluated.</p> |
| Soil Microorganisms | Exclude | | <p>While soil microorganisms play key roles in nutrient cycling within soils (as they are primary consumers of organic matter and 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 there are some soil quality benchmarks that utilize nutrient cycling parameters as the biological endpoints.</p> <p>There is high spatial and temporal variability in soils with respect to microbial community composition and tolerance to most contaminants. Reliable toxicity data are extremely limited and exposure-response relationships are unclear for most contaminants, especially those that can be utilized 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.</p> <p>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. Given these considerations, soil microorganisms were excluded as ROCs.</p> |

| Receptor Groups | Include/ Exclude | Surrogate Species for Receptor Group (if included) | Rationale for Inclusion/Exclusion |
|---------------------------|------------------|--|--|
| Herbivorous Large Mammals | Exclude | | <p>Herbivorous large mammals such as deer and moose could occur within the site (one fawn was noted within the AECs during filed work) but would not be anticipated to occur within the AECs to a large degree due to there being less habitat and food resources in these areas than adjacent areas. Given the large home range of large herbivorous mammals (e.g., white-tailed deer have a home range of 30 ha and moose have a home range of 460 ha; FCSAP, 2012b), and their large body weights (relative to other herbivores), the exposure potential to COPCs for these receptors is likely limited, especially given that the site has minimal vegetation in some areas.</p> <p>While exposure routes for this receptor group were not selected for evaluation (i.e., direct soil contact/soil ingestion and vegetation ingestion), the evaluation of these pathways for small mammal herbivores adequately captures potential worst case COPC exposures for large mammals. Small mammals have much smaller body weights and home/foraging range sizes than large mammals and would therefore incur much higher exposures and potential risks to site COPCs.</p> |
| Carnivorous Large Mammals | Exclude | | <p>While large carnivorous mammals occur in Nova Scotia, they generally have large home ranges. For example, the coyote home range is reported to be approximately 2,000 ha in Nova Scotia) and the bobcat home range is reported to be 100 ha to 10,000 ha) (NatureServe, 2023). While these species could be exposed to site chemicals via ingestion of prey from the site, given the limited food resources within the most contaminated areas of the study boundary, the large home ranges and body weights of carnivorous large mammals, their exposure potential to COPCs in media and biota at the site are likely limited. As such, carnivorous large mammals were excluded as ROCs.</p> <p>The evaluation of small mammal carnivores adequately represents or captures potential worst case COC exposures for large mammals, as small mammals have much smaller body weights and home/foraging range sizes than large mammals and would therefore incur much higher exposures to COPCs.</p> |

| Receptor Groups | Include/Exclude | Surrogate Species for Receptor Group (if included) | Rationale for Inclusion/Exclusion |
|---|-----------------|--|---|
| Carnivorous Birds | Exclude | | Carnivorous birds may be exposed to COPCs via the ingestion of prey items that occur or forage on the site, and incidental soil ingestion. The prey items on the site would consist primarily of herbivorous and/or insectivorous/invertevorous small mammals (and birds). The potential for exposure of carnivorous birds to site related COCs would be expected to limited due their large home ranges, and the limited available habitat/food resources with the contaminated areas of the study boundary for these birds, compared to other nearby areas. As such, carnivorous birds were excluded as ROCs. |
| Marine Pelagic and Benthic Aquatic Life | Exclude | | Marine surface water and aquatic receptors are not located on or near the site; thus, this receptor group was excluded as ROCs. |
| Waterfowl and Piscivorous Wildlife (bird and mammal piscivores) | Exclude | | No waterfowl or piscivores were observed on the site during the Phase II ESA (Dillon, 2022b) or subsequent site visits. While piscivorous wildlife could occur within the study boundary, their exposures to COPCs from site media and food items in these areas is likely limited. This is partly due to their relatively large home and foraging range sizes, the number of waterbodies in the area and varied diets for some species (e.g., fish, benthic invertebrates, aquatic vegetation). |

5.2.5.1 Size of Site and Potential Habitat

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 (US EPA, 1997; 1998; 1994; ASTM, 2014; MCP, 1996). Ideally, spatial scale issues are discussed at the onset of the ERA process. This can allow small sized sites that 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 (2014) states that the space or size of a contaminated site is directly related to the potential for ecological receptors to be exposed. Consideration of spatial scale can help focus an ERA on the issues or receptors of greatest ecological relevance, and/or provide a basis for determining that an ERA for a given site is not necessary for some or all receptors of interest. Similarly, the FCSAP (2012a) guidance identifies home range size, habitat suitability, and off-site habitat characteristics as factors that can affect the degree of exposures to site receptors.

ASTM (2014) suggests that for terrestrial environments (or sites), ecological habitat areas of less than two acres (equivalent to approximately 0.81 ha) are commonly considered too small to require an 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 these sites would not require 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, as long as the habitat and soil conditions are suitable. ASTM (2014) 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 land-based study boundary of the site is approximately 51 ha which is above the ASTM (2014) spatial criterion of 0.81 ha, indicating the site may be large enough support local populations of mammalian and avian receptors. This, however, does not consider the available habitat and food resources on the site, which are rather limited in many areas (i.e., within the central portion of the study boundary). There is less vegetation in the central areas of the site although trees and shrubs are more plentiful outside of this area and outside of the study boundary. In addition, watercourses and wetlands identified within the study boundary (**Section 3.3**) would not provide suitable habitat or foraging areas some terrestrial receptors.

5.2.5.2

Consideration of Habitat Quality and Habitat Preferences

ASTM (2016) provides guidance on using habitat quality and preference information as a means of determining if 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 (2016) states 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 (2016) 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. ERAs should only be conducted for sites 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 (2016) include the following:

- Consider the size of the site relative to receptor home/foraging/breeding range requirements; and
- 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, and breeding range size for the selected ROCs. Review of habitat preferences for these ROCs shows that the site has some conditions and ecological features that are consistent with their preferred habitat types (e.g., forested areas and open grassy fields); however, some areas within the central portion of the study boundary associated with the most contamination (i.e., within the AECs) provide limited suitable habitat for ROC due to the presence of waste rock piles and/or exposed rock, settling/tailings ponds, and limited vegetation. Therefore, while some areas within the study boundary do not have suitable habitat or food resources, other areas offer some suitable habitat conditions for various common mammal and avian receptors. As such, it is possible that local populations of these receptors could utilize the site for foraging, nesting, breeding, and other activities. This conclusion is generally supported by conducting N_s calculations 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 (2016). 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 (some alternate means of determining N_s are also provided in ASTM, 2016).

Equation 1

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.
- A_s = the area of the habitat subdivision (or portions of the site with potential habitat or foraging areas). In this case, the habitat subdivision of interest includes vegetated areas of the site which were assumed to be the area of the entire land based study boundary of approximately 51 ha.
- HR_s = the approximate home/foraging range size of the receptor (ha); based on literature regarding lowest reported home range sizes.

To examine whether the site could support populations of the mammalian ROC being examined, N_s values were calculated for the ROC in addition to other typical avian and mammalian species evaluated in ERA. N_s calculations for the receptors selected as ROC are presented below (**Table 5-3**). For all receptor species, the smallest identified HR_s value was taken from the FCSAP ERA guidance documentation (i.e., FCSAP, 2012b) or other resources if no FCSAP value was available. The A_s value used was approximately 51 ha which is the entire area within the study boundary.

Table 5-3: Estimated N_s Values for Selected Ecological Receptors at the Lake Enon Former Mill Site, Enon, Nova Scotia

| Receptor | A_s | HR_s | N_s |
|------------------------------|-------|----------|-------|
| Meadow vole | 51 | 0.348 ha | 146 |
| Masked shrew | | 0.6 ha | 85 |
| Dark-eyed junco ¹ | | 0.833 | 61 |
| Ruffed grouse | | 1 ha | 51 |
| Ermine ² | | 12 ha | 4 |
| Snowshoe hare | | 1.6 ha | 31 |
| White-tailed deer | | 30 ha | 1 |
| Black bear | | 300 ha | <1 |

Notes:

Home ranges from FCSAP, 2012b unless otherwise noted. The smallest home ranges provided were used for the N_s calculation with the exception of the meadow vole. The lowest home range for the meadow vole of 0.0069 ha would apply only if the habitat quality and food resources are exceptional, which they would not be on the site due to limited habitat and food resources. As such, the home range of 0.348 ha (FCSAP, 2012b) was used.

¹ Home range is female home range during nestling period (Reichard and Ketterson, 2012).

² Home range from the wildlife of Nova Scotia website which noted adult home ranges are 12 ha to 16 ha, with males having higher home ranges than females (Towers, 1980). The lower of these two home ranges was used for the N_s calculation. N_s values rounded down to nearest whole number.

Receptors include terrestrial receptors selected as ROC in this ERA in addition to other typical avian and mammalian species evaluated in ERAs.

While there is no set N_s value that represents the presence of a population for each receptor, the N_s values in **Table 5-3** suggest that a few individuals to potentially populations of receptors with smaller home ranges may occur on site. The N_s scores were calculated based on the total area of the site within the study boundary which is approximately 51 ha. If the areas of limited habitat and food resources/unsuitable habitat for ROC within the study boundary are considered, the N_s scores for each receptor would be lower; thus, these estimations are considered to be conservative.

While the N_s calculation is not without uncertainty and does not account for literature reports of actual measured population density and territoriality considerations, it provides a reasonable and often conservative indication of the numbers of individual organisms that may be expected to use a habitat subdivision on a given site. For the Lake Enon Former Mill site, these N_s values would be considered conservative given the limited habitat/food resources within some areas of the study boundary. As such, it would be unrealistic to assume that ROC presence and foraging occurs extensively within the study boundary, but not on the various adjacent areas that have equally suitable (or better) habitat conditions and food availability. For example, the central areas of the study boundary have less terrestrial vegetation than the adjacent areas which are mostly forested with more lush vegetation.

There is nothing to distinguish the site from other areas as offering unique or preferred habitat or food resources for any known ecological receptor population or community. In fact, areas outside of the

study boundary would offer better habitat and food resources than the habitat within the study boundary (particularly the central portion of the study boundary).

The calculated N_s scores support that the use of the site by small mammalian receptors, and to a lesser extent avian receptors, could potentially include populations. However, the N_s scores were calculated based on the total site area and did not exclude areas within the study boundary where there is no available habitat and/or limited food resources. Given much of the central portion of the study boundary has less vegetation and food resources and has either rock piles or settling/tailings ponds, it is unlikely that populations of carnivorous small mammals, larger mammals or birds could be supported within the study boundary (particularly in the most contaminated areas). However, the site would likely support populations of smaller mammals with smaller home ranges.

The N_s score for the junco indicates a potential local population of this receptor (i.e., $N_s = 61$). However, the home range used to estimate this N_s score was a home range of a female for breeding and nesting (Reichard and Ketterson, 2012). Non-breeding winter home range has been reported to be approximately 4 ha to 5 ha (10 to 12 acres; Sullivan, 1999; US DOA, 1991). When these home ranges are considered, the N_s scores are 12 and 10, respectively, which would not likely support a population. Additional support for the site not being able to support a population of junco, comes from a study in Minnesota, where the overall mean densities of the junco in national forest lands, ranged from 0.08 to 0.11 pairs per 40 ha (Minnesota Breeding Bird Atlas, 2023). Given the size of the study boundary is 51 ha, it is unlikely that there would be more than one or two nesting pairs of junco present.

Similarly, population densities of the ruffed grouse in North America are low and have been reported to average approximately 10 to 15 bird per square km (100 ha) (Collicutt, 2023). Given the size of the study boundary is approximately 51 ha, about half of that amount would be estimated. Based on these overall considerations, it is unlikely that populations of dark-eyed junco or ruffed grouse or other similar species would be supported within the study boundary.

5.2.5.3

Summary of ROC Selection

Based on the ROC selection procedure and related considerations described in the preceding sections, the receptors/receptor groups in **Table 5-4** were identified as ROCs that warranted further evaluation in the ERA.

Table 5-4: Receptors of Concern (ROC) Selected for Evaluation

| ROC | Surrogate | Comment |
|---------------------------|------------------------------|----------------------------------|
| Terrestrial vegetation | NA | Assessed at the community level |
| Soil invertebrates | NA | Assessed at the community level |
| Herbivorous small mammals | Meadow vole Snowshoe hare | Assessed at the population level |

| ROC | Surrogate | Comment |
|---|-----------------|---|
| Insectivorous/invertevorous small mammals | Masked shrew | Assessed at the population level |
| Herbivorous birds | Ruffed grouse | Assessed at the population level |
| Insectivorous/invertevorous birds | Dark-eyed junco | Assessed at the population level |
| Carnivorous small mammals | Ermine | Assessed at the population level |
| Amphibians | NA | Assessed at the community level |
| Freshwater aquatic life | NA | Assessed at the community level |
| SAR | Snapping turtle | Community level, qualitatively assessed |

Notes:

NA = not applicable

5.2.6 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; US EPA, 1998; FCSAP, 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 (FCSAP, 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 (FCSAP, 2012a). It is common practice in ERA that assessment endpoints do not express a direction or desired state (such as: increased, decreased, healthy, or sustainable).

Assessment endpoints may or may not be directly measurable (US EPA, 1998). For example, the abundance of songbirds 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 (FCSAP, 2012a). Measurement endpoints form the 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 (2007) defined measurement 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.”

A key consideration in the selection of measurement endpoints is how well a given 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.

FCSAP (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 involve the use of more than one measurement endpoint.

Essentially, measurement endpoints are tools, and LOEs are the way that 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 FCSAP (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 the literature, assuming that the concentration-response relationships between sites are similar;
- **Site-specific biological evidence** – Considers direct assessment of the site biological conditions; and
- **Indirect biological evidence** – Considers indirect assessment of biology, through extrapolation of knowledge obtained at other sites and from the literature.

Specific assessment and measurement endpoints and LOE were identified for the ROC that were selected for evaluation in the ERA and are presented in **Table 5-5**.

Table 5-5: Assessment and Measurement Endpoints and Lines of Evidence for the Selected ROCs

| Receptor of Concern (ROC) | Assessment Endpoint | Measurement Endpoints | Lines of Evidence (LOE) |
|---------------------------|--|--|---|
| Terrestrial vegetation | Survival, growth, and reproduction of vegetation communities | <p>Concentrations of chemicals in site soil and reference areas.</p> <p>Vegetation health-based soil quality benchmarks.</p> <p>Qualitative site survey of vegetation types and apparent health and similarity to surrounding sites.</p> | <p>Outcomes of the comparison of site soil chemical concentrations to vegetation health-based soil quality benchmarks and reference areas.</p> <p>Comparison of vegetation on site to vegetation present in the surrounding area and results of terrestrial habitat assessment.</p> |

| Receptor of Concern (ROC) | Assessment Endpoint | Measurement Endpoints | Lines of Evidence (LOE) |
|--|--|---|---|
| Terrestrial vegetation | | Consideration of the terrestrial habitat assessment. | Outcomes of visual vegetation observations and outcomes of the terrestrial habitat assessment. |
| Soil invertebrates | Survival and reproduction of soil invertebrate communities | Concentrations of chemicals in site soil and reference areas. Soil invertebrate health-based soil quality benchmarks. | Outcomes of the comparison of site soil chemical concentrations to soil invertebrate health-based soil quality benchmarks and reference areas. |
| Herbivorous small mammals (surrogates: meadow vole and snowshoe hare) | Survival, growth, reproduction, and abundance of populations | Concentrations of chemicals in site soils and reference areas. Mammalian health-based soil quality benchmarks. Modeled exposure and risk estimates for COCs in food items and soil. Measured tissue concentrations in wetland vegetation samples. Applicable mammalian TRVs. Spatial size and habitat quality/preference considerations for the site. N_s (the number of individuals of a given receptor species that are likely to inhabit any habitat subdivision on a site) values estimated for the site. Concentrations of COPCs in Wetland #3 vegetation tissue. | Outcomes of the comparison of site soil chemical concentrations to mammalian health-based soil quality benchmarks. Comparisons between estimated COC exposures and TRVs (i.e., ecological hazard quotients). Comparison of measured tissue concentrations versus modelled concentrations. Comparison of site spatial size to ASTM spatial criteria for ERA; comparison of receptor habitat preferences to site habitat conditions and habitat quality. Interpretation of N_s values with respect to likelihood for populations or individuals to utilize or occur on the site. Outcomes of the comparison of Wetland #3 vegetation tissue data to exposure point concentration used in exposure modelling. |
| Insectivorous or invertevorous small mammals (surrogate: masked shrew) | Survival, growth, reproduction, and abundance of populations | Concentrations of chemicals in site soils and reference areas. Mammalian health-based soil quality benchmarks. | Outcomes of the comparison of site soil chemical concentrations to mammalian health-based soil quality benchmarks. |

| Receptor of Concern (ROC) | Assessment Endpoint | Measurement Endpoints | Lines of Evidence (LOE) |
|---|---|---|---|
| <p>Insectivorous or invertevorous small mammals (surrogate: masked shrew)</p> | | <p>Modeled exposure and risk estimates for COCs in food items and soil.</p> <p>Applicable mammalian TRVs.</p> <p>Spatial size and habitat quality/preference considerations for the site.</p> <p>N_s (the number of individuals of a given receptor species that are likely to inhabit any habitat subdivision on a site) values estimated for the site.</p> | <p>Comparisons between estimated COC exposures and TRVs (i.e., ecological hazard quotients).</p> <p>Comparison of site spatial size to ASTM spatial criteria for ERA; comparison of receptor habitat preferences to site habitat conditions and habitat quality.</p> <p>Interpretation of N_s values with respect to likelihood for populations or individuals to utilize or occur on the site.</p> |
| <p>Herbivorous birds (surrogate: ruffed grouse)</p> | <p>Survival, growth, reproduction, and abundance of populations</p> | <p>Concentrations of chemicals in site soils and reference areas. Avian health-based soil quality benchmarks.</p> <p>Modeled exposure and risk estimates for COCs in food items and soil.</p> <p>Applicable avian TRVs.</p> <p>Measured tissue concentrations in wetland vegetation samples.</p> <p>Spatial size and habitat quality/preference considerations for the site.</p> <p>N_s (the number of individuals of a given receptor species that are likely to inhabit any habitat subdivision on a site) values estimated for the site.</p> <p>Concentrations of COPCs in Wetland #3 vegetation tissue.</p> | <p>Outcomes of the comparison of site soil chemical concentrations to avian health-based soil quality benchmarks.</p> <p>Comparisons between estimated COC exposures and TRVs (i.e., ecological hazard quotients).</p> <p>Comparison of measured tissue concentrations versus estimated tissue concentrations used in the ERA exposure model.</p> <p>Comparison of site spatial size to ASTM spatial criteria for ERA; comparison of receptor habitat preferences to site habitat conditions and habitat quality.</p> <p>Interpretation of N_s values with respect to likelihood for populations or individuals to utilize or occur on the site.</p> <p>Outcomes of the comparison of Wetland #3 vegetation tissue data to exposure point concentration used in exposure modelling.</p> |

| Receptor of Concern (ROC) | Assessment Endpoint | Measurement Endpoints | Lines of Evidence (LOE) |
|--|---|---|---|
| <p>Insectivorous or invertevorous birds (surrogate: dark-eyed junco)</p> | <p>Survival, growth, reproduction, and abundance of populations</p> | <p>Concentrations of chemicals in site soils and reference areas. Avian health-based soil quality benchmarks. Modeled exposure and risk estimates for COCs in food items and soil.</p> <p>Applicable avian TRVs.</p> <p>Spatial size and habitat quality/preference considerations for the site.</p> <p>N_s (the number of individuals of a given receptor species that are likely to inhabit any habitat subdivision on a site) values estimated for the site.</p> | <p>Outcomes of the comparison of site soil chemical concentrations to avian health-based soil quality benchmarks.</p> <p>Comparisons between estimated COC exposures and TRVs (i.e., ecological hazard quotients).</p> <p>Comparison of site spatial size to ASTM spatial criteria for ERA; comparison of receptor habitat preferences to site habitat conditions and habitat quality.</p> <p>Interpretation of N_s values with respect to likelihood for populations or individuals to utilize or occur on the site.</p> |
| <p>Carnivorous small mammals (surrogate: ermine)</p> | <p>Survival, growth, reproduction, and abundance of populations</p> | <p>Concentrations of chemicals in site soils and reference areas. Mammalian health-based soil quality benchmarks. Modeled exposure and risk estimates for COCs in food items and soil.</p> <p>Applicable mammalian TRVs.</p> <p>Spatial size and habitat quality/preference considerations for the site.</p> <p>N_s (the number of individuals of a given receptor species that are likely to inhabit any habitat subdivision on a site) values estimated for the site.</p> | <p>Outcomes of the comparison of site soil chemical concentrations to mammalian health-based soil quality benchmarks.</p> <p>Comparisons between estimated COC exposures and TRVs (i.e., ecological hazard quotients).</p> <p>Comparison of site spatial size to ASTM spatial criteria for ERA; comparison of receptor habitat preferences to site habitat conditions and habitat quality.</p> <p>Interpretation of N_s values with respect to likelihood for populations or individuals to utilize or occur on the site.</p> |

| Receptor of Concern (ROC) | Assessment Endpoint | Measurement Endpoints | Lines of Evidence (LOE) |
|--|---|--|--|
| Freshwater Aquatic Life | Survival, growth, reproduction, and abundance of populations. | <p>Concentrations of chemicals in site sediment, surface water and groundwater.</p> <p>Freshwater aquatic life health-based groundwater quality benchmarks.</p> <p>Consideration of the aquatic habitat assessment.</p> <p>Consideration of benthic community, and chemicals in aquatic vegetation and bivalve tissue.</p> | <p>Outcomes of the comparison of site sediment and surface water to freshwater aquatic life health-based sediment, surface water and groundwater benchmarks.</p> <p>Outcomes of the aquatic habitat assessment.</p> <p>Outcomes of the benthic community study, and aquatic vegetation and bivalve tissue concentrations.</p> |
| Amphibians | Survival, growth, reproduction, and abundance of populations. | <p>Concentrations of chemicals in site surface water and sediments and wetland areas.</p> <p>Amphibian toxicity data.</p> <p>Spatial size and habitat quality/preference considerations for the subject property.</p> <p>Consideration of the wetland habitat assessment.</p> | <p>Comparison of freshwater aquatic life guidelines and amphibian toxicity data to site concentrations.</p> <p>Comparison spatial size and habitat quality to surrounding areas, comparison of receptor habitat preferences to subject property habitat conditions and habitat quality.</p> <p>Outcomes of the wetland habitat assessment.</p> |
| SAR (i.e., Snapping Turtle (<i>Chelydra serpentina</i>)) | Survival, growth, reproduction, and abundance of individuals. | <p>Potential exposure routes</p> <p>Spatial size and habitat quality/preference considerations for the subject property.</p> <p>Consideration of the habitat assessment.</p> | <p>Qualitative outcomes of the exposure potential via the relevant exposure routes</p> <p>Comparison spatial size and habitat quality to surrounding areas, comparison of receptor habitat preferences to subject property habitat conditions and habitat quality.</p> <p>Outcomes of the habitat assessment.</p> |

Notes:

In the ERA context, the definition of a “population” can vary, and as such, very few ERA guidance documents define this term. FCSAP (2012a) states the following “From a pure biology perspective, an ecological population is defined as a group of organisms of a single species that interbreed and share a common habitat.” From a risk assessment perspective, however, this definition is too broad. 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 COCs 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.

5.2.7 Selection of Exposure Pathways and Routes

Ecological receptors can come into contact with chemicals in a variety of ways, depending on their daily activities, foraging and dietary behaviour and life history characteristics. The means by which an ecological receptor comes into contact with a chemical in an environmental medium are referred to as exposure pathways. The means by which a chemical enters the receptor from the environmental medium are referred to as exposure routes. If there are no possible exposure pathways that link ROCs to COPCs 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/identified receptor groups. Exposure pathways selected for the receptor groups in the ERA are provided below.

5.2.7.1 Terrestrial Vegetation and Soil Invertebrates

- Exposure of terrestrial plants to chemicals in soil is controlled by the root distribution in the soil profile, physicochemical characteristics of the soil (which determines the forms of the chemicals and their phytoavailability) and by interactions of the chemicals of interest with other chemicals present in soil (Suter II, 2007);
- Exposure of soil invertebrates to chemicals in soil is determined by depth of burrowing, the rate, and quantity of soil material ingested, activity patterns, soil physical-chemical characteristics, and interactions of the chemicals of interest with other chemicals in soil (Suter II, 2007); and
- The main exposure pathways and routes evaluated for terrestrial vegetation and soil invertebrates included were direct soil contact pathways, which include direct contact with the dermal coverings or integument of invertebrates, direct soil ingestion by invertebrates, and root uptake by plants.

5.2.7.2 Birds and Mammals

- For avian and 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 exposure pathways rarely require evaluation in ERAs (FCSAP, 2012a, Science Advisory Board of Contaminated Sites in British Columbia [SABCS], 2008; US EPA, 2003). This is considered to be the case in the current ERA as well, given that the ROCs have fur that would limit dermal contact of the COPC with skin, and given that the potential for significant soil/dust resuspension in local ambient air is low at the site for much of the year due to snow cover or frozen ground in the winter;
 - For the current ERA, it was assumed that herbivorous small mammalian ROCs are exposed to COCs in site soil via direct soil contact (soil ingestion) as well as the ingestion of vegetation;
 - For many small mammals, and many birds, it is common not to consume water from surface water bodies, but rather, to obtain the majority of their water requirements from puddles or droplets on vegetation surfaces, as well as from the consumption of food. Many small mammals and many bird species obtain their water requirements from consumed food items, wherein their physiological water needs are met via the metabolism of ingested food. For the small mammals and birds expected to be on the site, surface water ingestion would not be expected to be a pathway of concern;
 - The main exposure pathways and routes evaluated for herbivorous small mammals and birds were soil ingestion and ingestion of vegetation;
 - The main exposure pathways and routes evaluated for insectivorous/invertevorous small mammals and birds were soil ingestion and ingestion of soil invertebrates; and
 - The main exposure pathways and routes evaluated for carnivorous mammals were soil ingestion and ingestion of prey.

5.2.7.3

Freshwater Aquatic Life

The main routes and pathways by which these receptors may be potentially exposed to COPCs would include: direct contact of gills and other respiratory surfaces with water and sediments, sediment and water ingestion (including sediment pore water for benthos), and ingestion of aquatic prey species and other food items (i.e., detritus, plants, phytoplankton, zooplankton, macroinvertebrate fauna, and fish can all be food items for aquatic organisms depending on the number of trophic levels and aquatic food web structure). Sediment ingestion and pore water ingestion is more prevalent for bottom-dwelling or bottom-feeding species (Schoof, 2003). Aquatic plants are typically exposed to chemicals through root uptake of substances present in sediments and sediment pore water, and surface water uptake. For benthic species, the relative importance of whole (or bulk) sediment versus pore water exposures depends on the individual species, and their feeding and burrowing behavior. For example, pore water exposure may be insignificant to invertebrates that ingest sediment particles but may be an important exposure pathway for benthic organisms that burrow, and/or obtain their food by filter feeding.

5.2.7.4

Amphibians

Amphibians have a complex life history and can live in either aquatic or terrestrial habitats depending on life stage. As such, the evaluation of exposure pathways for amphibians is complicated. Generally, during embryo and larval life stages, amphibian exposures are primarily aquatic, while during juvenile and adult stages, exposures are primarily terrestrial. The main exposure pathways associated with life stages in the terrestrial areas of the site would include soil ingestion and direct contact and ingestion of terrestrial foods, while exposures in aquatic areas would include direct contact with sediment and surface water, incidental ingestion of soil/sediments and ingestion of aquatic foods.

5.2.7.5

Reptiles

The main exposure pathway for reptiles would be dietary exposures and dermal exposures. Depending on the reptile species, foods could be from terrestrial or aquatic habitats. Dermal exposures via soils and surface water (depending on the species and chemicals present on site) may also be important.

5.2.7.6

Exposure Pathway Summary

With respect to the assessment and measurement endpoints and LOEs noted in **Table 5-5** and the exposure pathways and routes selected for the ROC, there were some inherent conservative assumptions within the ERA that must be acknowledged. It is assumed that the ROC forage entirely in areas of the site that contain impacted soils and that their diet consists entirely of food from within the study boundary. While these are typical assumptions for an ERA of a contaminated site, they are considered conservative, particularly given the limited habitat and food resources on some areas of the site compared to the surrounding area.

For the receptor groups selected for evaluation in the current ERA, the following information on exposure pathways applies. Exposure pathways and routes for each receptor group are summarized in **Table 5-6**.

Table 5-6: Summary of Exposure Pathways and Routes Selected for the Receptor Groups

| Receptor Group | Exposure Pathway Selected for Evaluation |
|--|---|
| Terrestrial vegetation | <ul style="list-style-type: none"> • Direct contact with soils • Root uptake |
| Soil invertebrates | <ul style="list-style-type: none"> • Direct soil ingestion • Direct contact with dermal coverings or integument |
| Herbivorous small mammals | <ul style="list-style-type: none"> • Incidental soil ingestion • Vegetation ingestion |
| Insectivorous or invertevorous small mammals | <ul style="list-style-type: none"> • Incidental soil ingestion • Ingestion of soil invertebrates |

| Receptor Group | Exposure Pathway Selected for Evaluation |
|--------------------------------------|---|
| Herbivorous birds | <ul style="list-style-type: none"> • Soil ingestion • Vegetation ingestion |
| Insectivorous or invertevorous birds | <ul style="list-style-type: none"> • Incidental soil ingestion • Ingestion of soil invertebrates |
| Carnivorous small mammals | <ul style="list-style-type: none"> • Incidental soil ingestion • Ingestion of prey |
| Freshwater Aquatic Life | <ul style="list-style-type: none"> • Direct contact with sediment • Direct contact with surface water |
| Amphibians | <ul style="list-style-type: none"> • Soil and sediment ingestion • Direct contact with soil, sediments, and surface water • Ingestion of aquatic and terrestrial foods |
| Reptiles (snapping turtle) | <ul style="list-style-type: none"> • Soil and sediment ingestion • Direct contact with soil, sediments, and surface water • Ingestion of aquatic and terrestrial foods |

5.2.8 Identification of COPCs

The goal of COC selection is to identify those chemicals measured in site/property media that may pose a potential risk to ecological health. Details regarding the COC identification process for the ERA (including the data and statistics, benchmarks that were used, COC screening tables and screening outcomes) are provided in the following sections and **Appendix J**.

Similar to the COC identification process for the HHRA, the identification of COCs for the ERA also involved a sequential process, as follows.

1. Comparison of the maximum measured concentrations of contaminants in site soil to applicable regulatory ecological health-based benchmarks and available local background data;
2. Chemicals with a maximum concentration that exceeded the applicable ecological health-based benchmark and were above background concentrations, were identified as initial COPCs;
3. These initial COPCs were further evaluated by determining the frequency, magnitude, and spatial extent of relevant media exceedances from the site above the applicable receptor-specific ecological health-based benchmarks, where available. For mobile receptors (e.g., mammals and birds), the initial COPCs were further screened against a 90th percentile soil concentrations as per FCSAP (2012a);
4. Further considerations (where/if deemed necessary and relevant), such as:
 - Essential nutrient status of certain inorganic elements;
 - Ubiquity of elements in soil, water or sediments;
 - Statistical relationships;

- Local geology;
 - Spatial distribution of soil chemical concentrations;
 - Fate and behaviour properties (including potential to bioaccumulation and biomagnification);
 - Speciation of chemicals in site media; and
5. Final COCs are selected considering all other above, where relevant.

COC identification in site media was guided by the following considerations:

- Substances that were non-detectable (below RDLs) in all site media samples were excluded from further consideration unless the RDL was above the applicable benchmark value;
- All site data were considered, including field duplicate samples;
- Soil concentrations of analytes were assumed to be accessible for ecological receptor exposure regardless of location and considered soils up to 1.5 m deep;
- The more conservative of coarse or fine-grained soil benchmarks were used based on the particle size analysis (Dillon, 2022b);
- With respect to sediment benchmarks, only probable effect level (PEL) and similarly derived benchmarks were applied (i.e., examples of sediment benchmarks similar to a PEL are effects range median (ERM) and severe effect levels (SEL)). It has become well established in the scientific and regulatory literature over the years that exceedances over PELs and similar benchmarks are often a better or more reliable indicator of potential sediment toxicity (or an increased likelihood for benthic community impairment) than low effect level sediment benchmark exceedances (such as ISQGs or interim sediment quality guidelines). The past experience of Dillon personnel in numerous ERAs of contaminated sediments has shown that exceedance of low effect level benchmarks often does not correlate well with other endpoints that are commonly evaluated in aquatic risk assessments (e.g., sediment bioassay results and benthic community parameters). In practice, exceedance of the PEL (and similar values) is the more realistic indicator of a potential for population or community level adverse effects for benthic organisms and is also typically the most suitable basis for determining when/if further action or assessment is necessary. The use of the PEL is well supported in the literature (e.g., NOAA, 1999; Long et al., 1998; MDEP, 2006a,b). There is also considerable precedent for reliance on PEL exceedances as a means of determining the need for further assessment or action when evaluating contaminated sediments. For example, there are numerous papers in the scientific literature that calculate PEL quotients (not ISQG, or other low effect level quotients) when evaluating sediment contamination and ranking aquatic contaminated sites for further action or no action. This approach is also common to a number of sediment assessment frameworks used within Canada (e.g., FCSAP Aquatic Sites Classification System);
- Chemicals having a maximum concentration which exceeded the applicable ecological benchmark value and available local background data were retained for further assessment as COPCs. It is a general assumption in all risk assessment studies that there is a low to negligible likelihood for potential adverse effects when maximum media chemical concentrations are

below such benchmarks. Media chemistry data that meets these conditions are typically considered to require no further evaluation or action; and

- Chemicals without environmental health-based benchmarks were carried forward for further discussion.

The COC identification processes used are consistent with site and risk assessment procedures and help to distinguish natural occurring concentrations from those that have been influenced by anthropogenic activities.

With respect to the regulatory context for the ERA, the site is located on land owned by the NSDNRR, and as such the NS CSR guidelines apply and were preferentially used in the COPC screening. As part of the NS Contaminated Sites protocol, Tier I EQS and Tier II PSS for soil, groundwater, surface water and sediment have been selected for use in NS. Links to these EQS and PSS are provided on the NS CSR website (<https://novascotia.ca/nse/contaminatedsites/protocols.asp>). While the ecological health-based PSS for commercial soils were not found on the NS CSR website, these values have been provided by Atlantic RBCA and have been adopted by the province. Except for petroleum hydrocarbons and select chlorinated volatile organic compounds (CVOCs), the Atlantic RBCA EQS/PSS values are adopted values that have been derived by CCME and other regulatory authorities in Canadian or international jurisdictions. For contaminants other than PHCs and CVOCs, the general hierarchy used to select EQS/PSS is outlined in the Environmental Quality Standards Rationale and Guidance Document (2021). The PSS were developed by source agencies and departments outside of Nova Scotia such as the CCME, AEP and MECP. The original guideline sources were reviewed to ensure the NS PSS guideline referenced the most recent applicable guideline from the original source. Where no NS PSS was identified, other provincial or federal regulatory benchmarks were used, where available. In the absence of Canadian guidelines, the US EPA EcoSSL (ecological soil screening levels) were applied. As previously noted, since the site is zoned as commercial, commercial land use guidelines were selected for screening purposes for the ERA.

For terrestrial vegetation and soil invertebrates, freshwater aquatic life and amphibians, the focus of an ERA is on community level effects, and for birds and mammals, the focus is on population level effects (with the exception of SAR). As such, an exceedance of a maximum concentration over a guideline is not indicative of potential risks. Assuming that exposures occur only to the maximum soil, surface water or sediment concentration is highly conservative and unrealistic. For mobile receptors, FCSAP (2013a) guidance states that if maximum concentrations are exceeded, then consideration should be given to summary statistics (e.g., compare the 95th upper confidence limit on the mean [UCLM] or the 90th percentile of the concentrations to the guideline) for screening when data sizes are reasonable (e.g., 10 or more samples in the area used by a receptor), as these metrics are more realistic for determining true COPCs. Therefore, in addition to the maximum media concentrations, the 90th percentile concentrations have also been considered in some cases. In addition, the frequency, magnitude, and spatial extent of

exceedance of Study Area soil, surface water or sediment data over applicable ecological health-based benchmarks was also considered before final selection of a chemical as a COPC.

An initial list of COPCs was identified by comparing maximum concentrations in soils within the study boundary soil, surface water, sediments and groundwater to ecological health-based guidelines and local background data. These initial COPCs were then considered further to select the final COCs for the ERA. **Appendix J** provides results of the COPC screening. A summary of analytical data used in the COPC screening is provided in **Appendix E**. Laboratory certificates for these data are provided in **Appendix C** and **Appendix F**. Results of the soil screening were used to identify COCs for terrestrial vegetation, soil invertebrates and small mammals (herbivorous, insectivorous and carnivorous). Results of the sediment and surface water screening were used to identify COCs for freshwater aquatic life and amphibians.

Published regulatory benchmarks are generic values, with no consideration given to site-specific populations/communities or media conditions that influence bioavailability. Also, they are developed by regulatory agencies to be intentionally conservative and protective. Exceedance of these values does not necessarily imply there is a risk of adverse effects; rather, it suggests that further evaluation is warranted (such as further ecological risk assessment or data collection). This is especially true for metals, many of which have essential nutritional and physiological roles in terrestrial biota. When interpreting screening level exceedances, it is also important to consider the body of literature regarding acclimation and adaptation of terrestrial organisms to metals and other substances in soil and/or food items. For example, it is well established that populations chronically exposed to metals often show an enhanced tolerance relative to populations with no, or lower exposure (Kapustka et al., 2004). This increased tolerance can be due to either acclimation (shifting of tolerance within the genetically defined limit of the organism) or adaptation (modification of the limits of an organism through changes in heritable genetic material) (International Council on Mining and Metals [ICMM], 2007). Increased metal/metalloid tolerance has been documented for many species of terrestrial and aquatic plants, animals, and microbes. For the most part, acclimation and/or adaptation have been demonstrated primarily at the population level, but studies of pollution-induced community tolerance have also documented these phenomena at the community level at various metals-contaminated sites. Acclimation and adaptation to metals is described further in Kapustka et al., (2004), ICMM (2007), Chapman (2008), and Newman and Clements (2008).

5.2.8.1

COPCs in Site Soil

FCSAP (2012a) states that all data in the top 1.5 m of soil should be considered relevant for screening COPCs for an ERA. However, soil depths greater than 0.3 m bgs are generally considered not to be available for direct contact or ingestion from ecological receptors (Anderson et al., 2010; Suter et al., 2000; Suter, 2007). Given some of the soils were wet and on ground that was marsh-like, it was assumed soil exposures could occur at deeper depths. As such, all soil samples collected within the top 1.5 m were screened against direct contact soil guidelines. Sample locations are presented on **Figure 3**,

Appendix A. Site photographs are presented in **Appendix B.** All soil data used in the current assessment are presented in **Appendix E.**

For a number of regulatory jurisdictions, ecological soil quality benchmarks have been derived for specific receptor types such as terrestrial vegetation, soil invertebrates, mammals and birds. In these cases, the lowest benchmark provided was selected for COC identification purposes.

The specific regulatory ecological health-based soil quality benchmarks that were used to identify COPCs in site soil are briefly described below. Based on soil grain size data (Dillon, 2022b), the site was considered to have coarse and fine-grained soils and as such, soil data were compared to the lower of coarse- and fine-grained soil guidelines, where applicable.

The specific source of the guideline used in the COPC screening of soils are identified in the ecological health screening tables (**Table J-1, Appendix J**).

Canadian Council of Ministers of the Environment (CCME)

The CCME Canadian soil quality guidelines developed for the protection of ecological health (SoQG_E), commercial land use (coarse soils), were preferentially applied. The lowest of the ecological soil contact or soil and food ingestion guidelines were used. Where the CCME soil guidelines defer to the older CCME (1991) soil remediation criteria (which are not identified as being either human or ecological health-based), the benchmarks from the next preferred jurisdiction were selected rather than using the outdated CCME (1991) value(s), where possible. If no other guideline was available from another jurisdiction, the CCME (1991) value was used. The most up to date CCME guidelines are available from on the online CCME summary tables and associated fact sheets (<http://st-ts.ccme.ca/en/index.html>; searched May 2022).

CCME Canada Wide Standards for PHCs in Soils (CCME, 2008)

The Canada Wide Standards (CWS) for PHCs for commercial land use and lower of fine and coarse-grained soils applicable for ecological health (ecological soil contact) were selected for comparison purposes for PHC F1 to F4 (CCME, 2008). Details on the basis and derivation of the CWS human health SoQGs are provided in CCME (2008).

Alberta Environment and Parks (AEP, 2022)

The lower of the direct soil contact, livestock soil and food ingestion, and wildlife soil and food ingestion ecological health-based soil guidelines for commercial land use and coarse-grained soils were applied. These AEP guidelines are developed in a very similar manner to, and are protective of, the same ecological receptor types as the CCME SoQG_E. Details on soil guideline basis and derivation are provided in AEP (2022).

Ontario Ministry of the Environment, Conservation, and Parks (MECP, 2011)

The lowest of the relevant MECP ecological health-based soil component values (i.e., soil component values for plants and soil organisms, soil component values for mammals and birds) were applied. The specific soil component values that were used are from Table 3 in MECP (2011), for conditions of non-potable water, institutional land use, and coarse-textured soils. The derivation of these values is described in MECP (2011).

Atlantic Risk-Based Corrective Action (RBCA)

The Atlantic RBCA (2022) soil quality guidelines for petroleum hydrocarbons for commercial sites were used for PHCs where applicable.

United States Environmental Protection Agency Ecological Soil Screening Levels (US EPA)

Where no Canadian value was available, the lowest relevant United States Environmental Protection Agency Ecological Regional Screening Level (US EPA EcoSSL) was used in the screening. The lower of plant, soil invertebrate, avian, and mammalian EcoSSLs were applied. The EcoSSL background and derivation documents can be found at <https://www.epa.gov/chemical-research/interim-ecological-soil-screening-level-documentsat>.

Oregon Department of Environmental Quality Risk-Based Concentrations (2020)

The Oregon Department of Environmental Quality Risk-Based Concentrations for Plants, Invertebrates, and Wildlife Exposed to Soil were applied where no other guideline was available. These values were obtained from the Los Alamos National Library (LANL) ECORISK Database (LA-UR-17-26376, Los Alamos National Laboratory, Los Alamos, New Mexico; LANL 2017, 602538). The lower of the herbivore, omnivore, or mixed diet exposure scenarios was applied. Background and derivation documents can be found at <https://www.oregon.gov/deq/FilterDocs/EcoRiskTablesAppendices.pdf>.

Results of Ecological Health-Based Soil Screening

Site soil screening results are provided in **Table 5-7** and **Table 5-8** and discussed in the following sections. Details of the ecological screening for site soil are provided in **Table J-1**, **Appendix J** and **Figures 10 to 14 (Appendix A)** show the locations of guideline exceedances for metals in soil. Results of the soil screening were used to identify COCs for terrestrial vegetation, soil invertebrates, and mammalian and avian ROC.

Chemicals that were not detected in any of the analyzed soil samples and were not associated with current and historical activities at the site were excluded from further assessment. For metals, antimony and boron were not detected in soil samples and were excluded from further assessment.

The primary contaminant group of concern at the site is metals; however, soil data were also available for the following contaminant groups: PAHs, PHCs, VOCs, sVOCs, and phenolics (see **Appendix E**). Analytes in these contaminant groups were all non-detect with the exception of the analytes listed

below, which were detected at low frequency and below their applicable ecological health-based guideline, where available. For completeness, the RDL for parameters with concentrations < the RDL were cross-referenced with the appropriate ecological-health based guideline, and all RDLs were below the applicable guideline; therefore, none of the non-detected parameters were carried forward in the ERA.

No guidelines were available for aluminium, bismuth, iron, and strontium. The maximum aluminium concentration was less than the maximum background concentration and was not considered further. Bismuth was detected in 3 of 103 samples at concentrations slightly over the detection limit and the maximum rubidium concentration was less than within the study boundary; as such, bismuth and rubidium were not carried forward for further evaluation.

Strontium was carried forward for further assessment as it was elevated across the site relative to background concentrations and is expected to be related to previous site activities.

Iron and manganese were previously discussed in the HHRA (**Section 4.2.4.1**) and are considered to be related to local background conditions and not site related. As such, these chemicals were not carried forward for further assessment.

Table 5-7: Ecological Screening (Soil Metal COPC Identification) for the Lake Enon Former Mill Site

| Chemical | Ecological Soil Quality Benchmark (SoQGE) (mg/kg) | Maximum Background Concentration (mg/kg) | Maximum Measured Concentration (mg/kg) | Exceeds Guideline and Background? (Y/N) |
|--------------------------|---|--|--|---|
| Metals | | | | |
| Aluminium | NGA | 38,000 | 31,000 | NA |
| Arsenic | 26 ^a | 6.2 | 15 | N |
| Barium | 2000 ^b | 1100 | 1,500 | N |
| Beryllium | 8 ^b | 1.6 | 2 | N |
| Bismuth | NGA | <2.0 | 2.9 | NA |
| Cadmium | 1.9 ^c | 2.6 | 25 | Y |
| Chromium (Total, III+VI) | 87 ^a | 21 | 34 | N |
| Cobalt | 180 ^c | 13 | 270 | Y |
| Copper | 91 ^a | 42 | 120 | Y |
| Iron | NGA | 31,000 | 100,000 | NA |
| Lead | 32 ^c | 170 | 30,000 | Y |
| Manganese | 220 ^d | 5,200 | 8,900 | Y |
| Mercury | 20 ^c | 0.56 | 0.39 | N |
| Molybdenum | 40 ^b | 2.7 | 12 | N |
| Nickel | 89 ^a | 18 | 69 | N |

| Chemical | Ecological Soil Quality Benchmark (SoQGE) (mg/kg) | Maximum Background Concentration (mg/kg) | Maximum Measured Concentration (mg/kg) | Exceeds Guideline and Background? (Y/N) |
|---|---|--|--|---|
| Selenium | 2.9 ^a | 14 | 3.5 | N |
| Rubidium | NGA | 29 | 28 | NA |
| Lithium | 350 ^e | 41 | 35 | N |
| Silver | 40 ^b | 1.3 | 4.3 | N |
| Strontium | 950 ^e | 470 | 27,000 | Y |
| Thallium | 3.6 ^a | 0.42 | 0.52 | N |
| Tin | 300 ^b | 1.7 | 5.5 | N |
| Uranium | 33 ^c | 3.9 | 11 | N |
| Vanadium | 18 ^c | 43 | 73 | Y |
| Zinc | 340 ^c | 500 | 3,600 | Y |
| Polycyclic Aromatic Hydrocarbons (PAH) | | | | |
| 1- and 2-methylnaphthalene ^f | NGA | - | 0.1 | NA |
| Fluoranthene ^f | 180 ^a | - | 0.06 | N |
| Phenanthrene ^f | 12 ^c | - | 0.031 | N |
| Petroleum Hydrocarbons (PHC) | | | | |
| Modified TPH ^g | 1700 ⁱ | - | 2600 | Y |
| PHC F2 (>C10-C16) ^h | 260 ^j | - | 17 | N |
| PHC F3 (>C16-C34) ^h | 1700 ^j | - | 950 | N |
| PHC F4 (>C34-C50) ^h | 3300 ^j | - | 519 | N |

Notes:

N = 103 for all metals except lead for which N = 114.

Where applicable, guidelines were taken from Atlantic RBCA Ecological Tier II Pathway-Specific Standards for Soil - Commercial/Industrial Land Use (July, 2022). The original source is indicated below.

NGA = no guideline available; **BOLDED** values indicate concentrations above applicable guideline and background; *Italicized* values indicate exceedances over the guideline, but not background; < = concentration was less than reportable detection limit (RDL), value provided is the RDL.

- a. Guideline is from CCME Factsheets. SoQGE (soil quality guideline ecological health); commercial land use; lower of direct contact pathways; coarse soils.
- b. Alberta Environment Protection (2019) - surface soil remediation guideline for commercial/industrial land use for direct contact pathway; ecological; coarse-grained soils.
- c. MECP (2011) Table 3 Soil Components - full depth, non-potable water - institutional land use; coarse soils; lowest of mammals & birds and plants & soil organisms
- d. US EPA EcoSoil Screening Level (EcoSSL) for manganese (US EPA, 2007). Lower of plant (220 mg/kg), soil invertebrate (450 mg/kg), avian (4300 mg/kg), and mammalian (4000 mg/kg) EcoSSLs.
https://www.epa.gov/sites/default/files/2015-09/documents/eco-ssl_manganese.pdf
- e. Oregon Department of Environmental Quality (2020) Risk Based Concentrations for Plants, Invertebrates and Wildlife Exposed to Soil. Derived from the LANL (Los Alamos National Laboratory), September 2017. "ECORISK Database (Release 4.1)", LA-UR-17-26376, Los Alamos National Laboratory, Los Alamos, New Mexico. (LANL 2017, 602538).

| Chemical | Ecological Soil Quality Benchmark (SoQGE) (mg/kg) | Maximum Background Concentration (mg/kg) | Maximum Measured Concentration (mg/kg) | Exceeds Guideline and Background? (Y/N) |
|----------|---|--|--|---|
|----------|---|--|--|---|

Lowest of herbivore, omnivore, or mixed diet exposure scenarios.

<https://www.oregon.gov/deq/FilterDocs/EcoRiskTablesAppendices.pdf>. Selected benchmark is the lower of the ground feeding and top consumer mammal risk based concentrations. No vegetation, soil invertebrate or avian values were available.

f. N = 23

g. N = 8

h. N = 1; Of the 6 samples analyzed for BTEX/TPH, one sample, SSSS12, did not reach baseline. Therefore, F4 analysis was requested which includes the F2-F4 fractions. As such, there is only one sample result for PHC F2 to F4.

i. Atlantic PIRI, 2022; Guideline of 1,700 mg/kg is for the F3 fraction (>C16-C34), which corresponds most closely to the TPH resemblance reported by the laboratory in the current field program (likely lube oil fraction).

j. Canada Wide Standard (CWS; CCME, 2008) for ecological direct contact (ingestion and dermal contact), commercial land use, lower of coarse and fine-grained soils.

Cadmium, cobalt, copper, lead, strontium, vanadium, and zinc exceeded both the soil quality guideline and background and were carried forward as initial COPCs for further assessment.

As for analytes other than metals, the following parameters were detected in soil in at least one sample are presented in **Table 5-7.**, with analytical results for the parameters or all samples being in **Appendix E**. All measured concentrations were below their applicable ecological-health based guideline, and were therefore excluded from further assessment. The only exception to this is modified TPH which is discussed below.

Although the maximum reported concentration of modified TPH of 2,600 mg/kg exceeds the ARBCA guideline of 1,700 mg/kg (for PHC F3 for coarse soils) by approximately 1.5-fold, the exceedance was only observed in one of the 8 samples collected, with the other samples being low in comparison (i.e., 21 to 99 mg/kg) (**Appendix E**). PHCs are readily metabolized by vertebrates and do not accumulate in tissues. In addition, PHCs are not readily absorbed into and accumulated into plants (CCME, 2008). Given these considerations and the low frequency and degree of exceedance, modified TPH was not carried forward.

No ecological-health based soil quality benchmarks were available for the following parameters, but they were detected in at least one sample:

- 1-Methylnaphthalene (detected in 3 of 23 samples; max. concentration 0.05 mg/kg); and
- 2-Methylnaphthalene (detected in 4 of 23 samples; max. concentration 0.05 mg/kg).

Given that the majority of PAHs were not detected, and detectable concentrations of 1- and 2-methylnaphthalene were reported in only three and four samples, respectively, it is unlikely that these parameters would be of concern to ecological receptors at the site. Additionally, the US EPA EcoSSL for Low Molecular Weight PAHs (LMWPAHs) of 29 mg/kg for soil invertebrates and 100 mg/kg for mammals

are at least two orders of magnitude higher than the highest observed concentrations of these parameters (0.020 and 0.04 mg/kg for 1- and 2-methylnaphthalene, respectively). Given that most PAHs were not detected, there is a negligible likelihood that the sum of LMWPAHs would exceed the guideline. Therefore, 1- and 2-methylnaphthalene were not carried forward for further assessment. Based on the comparisons presented in **Table 5-7** and **Table J-1, Appendix J** and data provided in **Appendix E**, the following chemicals were carried forward for further evaluation as initial COPCs (on the basis of maximum site soil concentrations exceeding applicable ecological soil quality benchmarks and/or local background concentrations; number of samples that exceed provided in parentheses):

Initial COPCs

- Cadmium (33 of 103);
- Cobalt (1 of 103);
- Copper (3 of 103);
- Lead (74 of 114);
- Strontium (39 of 103);
- Vanadium (5 of 103); and
- Zinc (33 of 103).

As previously mentioned, the ecological health-based benchmarks selected in the initial soil screening were the lowest available benchmark for any receptor. The chemicals identified as initial COPCs were then carried forward for a receptor-specific secondary screening (i.e., separate screening conducted for soil invertebrates and vegetation, and mammals and avian ROC) (**Table 5-8**). The soil quality guidelines specific to vegetation and soil invertebrates for direct soil contact exposure pathways were compared to the maximum site soil chemical concentrations as these receptors are immobile. The soil quality guidelines specific to mammalian and avian receptors for soil and food ingestion exposure pathways, were compared to the 90th percentile site soil chemical concentrations. This is in keeping with FCSAP (2012a) guidance, which indicates that a 90th percentile soil concentration can be used for screening mobile receptors if the sample size is >10 and a site is adequately characterized. This site had >10 samples and the site has been adequately characterized; as such, the 90th percentile soil concentrations were used in the secondary screening. Soil concentrations were also compared to the background soil concentration range (**Table 5-8**).

Table 5-8: Comparison of Lake Enon Former Mill Site Soil Chemistry Data for Initial COPCs to Receptor-Specific Ecological Health-Based Soil Quality Benchmarks and Local Background

| Chemical | Number of Samples | # Detectable Samples | Max Concentration (mg/kg) | 90th Percentile (mg/kg) ^d | Background Range (mg/kg) ^c | Ecological Health-Based Soil Quality Benchmarks by Receptor Group (mg/kg) | | Number of Samples Exceeding Ecological Soil Quality Benchmarks and Background | | Frequency of Exceedance Over Ecological Soil Quality Benchmarks and Background (%) | |
|-----------|-------------------|----------------------|---------------------------|--------------------------------------|---------------------------------------|---|----------------------------|---|----------------------------|--|----------------------------|
| | | | | | | Terrestrial Vegetation/ Soil Organisms ^a | Birds/Mammals ^b | Terrestrial Vegetation/ Soil Organisms ^a | Birds/Mammals ^b | Terrestrial Vegetation/ Soil Organisms ^a | Birds/Mammals ^b |
| Cadmium | 103 | 91 | 34 | 5.9 | <0.3 - 2.6 | 22 | 1.9 | 1 | 33 | 1% | 32% |
| Cobalt | 103 | 100 | 270 | 12.8 | <1.0 - 13 | 300 ^e | 180 | 0 | 1 | 0% | 1% |
| Copper | 103 | 99 | 120 | 47.8 | <2.0 - 42 | 91 | 3,100 | 3 | 0 | 3% | 0% |
| Lead | 114 | 114 | 30,000 | 3,440 | 7 - 170 | 600 | 32 | 32 | 74 | 28% | 65% |
| Strontium | 103 | 102 | 27,000 | 10,800 | <5 - 470 | NGA | 950/190,000 ^h | NA | 39/0 | NA | 38%/0% |
| Vanadium | 103 | 103 | 73 | 40 | 7.8 - 43 | 130 | 18 | 0 | 3 | 0% | 3% |
| Zinc | 103 | 103 | 3,600 | 824 | <5.0 - 500 | 450 | 960 ⁱ | 29 | 6 | 28% | 6% |

Notes:

Bold indicates the maximum concentration exceeded the vegetation/soil organism guideline; or the 90th percentile exceeds the bird/mammal guideline and background range.

^a CCME SoQGE soil contact guidelines unless otherwise noted; commercial land use; lower of coarse or fine-grained soil.

^b MECP (2011) Table 3 Soil Components - Full Depth, Non-Potable Water - Institutional land use; Coarse soils; Lower of Mammals & Birds Soil Components.

^c Range of soil samples collected by Dillon (2022, 2023) within the property that are not considered to be impacted and assumed to represent local background.

^d For 90th percentile calculations, non-detectable concentrations were assumed to equal the detection limit and the higher of a sample and its corresponding field duplicate were used in the calculation. Values rounded to 2 significant figures.

^e Alberta Environment and Parks (2019) - Ecological direct soil contact; Commercial land use

^f US EPA EcoSoil Screening Level (EcoSSL) for manganese (US EPA, 2007). Lower of plant (220 mg/kg) and soil invertebrate (450 mg/kg) EcoSSLs. https://www.epa.gov/sites/default/files/2015-09/documents/eco-ssl_manganese.pdf

^g US EPA EcoSoil Screening Level (EcoSSL) for manganese (US EPA, 2007). Lower of avian (4300 mg/kg) and mammalian (4000 mg/kg) EcoSSLs. https://www.epa.gov/sites/default/files/2015-09/documents/eco-ssl_manganese.pdf

^h Oregon Department of Environmental Quality (2020) Risk Based Concentrations for Plants, Invertebrates and Wildlife Exposed to Soil. Values provided represent ground feeding mammals (950 mg/kg) and top consumer mammals (190,000 mg/kg). No benchmarks were available for birds. Derived from the LANL (Los Alamos National Laboratory), September 2017. "ECORISK Database (Release 4.1)", LA-UR-17-26376, Los Alamos National Laboratory, Los Alamos, New Mexico. (LANL 2017, 602538). Lowest of herbivore, omnivore, or mixed diet exposure scenarios. <https://www.oregon.gov/deq/FilterDocs/EcoRiskTablesAppendices.pdf>

ⁱ The CCME (2018) developed updated zinc guidelines which included a commercial soil contact guideline of 450 mg/kg and an agricultural soil and food ingestion guideline of 960 mg/kg. The CCME (2018) agricultural soil and food ingestion guideline was selected for the evaluation of birds and mammals.

Table 5-8 compares maximum site soil chemical concentrations (for the initial COPCs) to vegetation and soil invertebrate-specific ecological health-based soil guidelines and compares 90th percentile soil chemical concentrations across the entire site for the initial COPCs (n = 103 for all metals except 114 for lead) to mammal and avian-specific ecological health-based soil guidelines. The maximum and 90th percentile concentrations were also compared to background soil concentrations where available. Chemicals that exceeded both the applicable guideline and available background concentrations were assessed further. **Table 5-8** also provides the frequency of soil guideline exceedance.

For the initial COPCs identified in **Table 5-7**, a significance level of 5% was used for both frequency of detection and frequency of exceedance. While there is no consistent regulatory guidance on a significance level or cut-off value to use when considering the frequency of detection, and/or exceedance over benchmarks in a COPC identification process, a value of 5% has some regulatory precedent within the U.S. Superfund program and as per the U.S. Navy HHRA Guidance document (US Navy, 2008). The US EPA Superfund Program has used a 5% cut-off value for detection frequency to eliminate chemicals from further consideration since the early 1990s, and the U.S. Navy (2008) HHRA guidance also supports the use of a 5% cut-off value when considering detection and benchmark exceedance frequency. Thus, the initial COPCs in **Table 5-7** carried forward for the receptor specific benchmark comparison that displayed a frequency of detection, and/or a frequency of exceedance over the applicable soil benchmark of less than 5%, were excluded from further consideration in the HHRA. If the frequency of detection was equal to or greater than 5% for the carried forward substances, they were then evaluated on the basis of their frequency of exceedance over the applicable soil quality benchmark. In addition, the degree of benchmark exceedance was also considered. If the frequency of exceedance over the soil benchmark was equal to or greater than 5%, then the substance was carried forward into the next step of the COPC identification process. Also, a 5% significance level is consistent with the default alpha value used in most statistical calculations with environmental data sets. This approach in COPC identification procedures have been applied for at least twenty years in the numerous ERAs and HHRAs conducted by Dillon, and the approach has been accepted by the applicable federal and provincial regulatory authorities.

For vegetation and soil invertebrate ROC, the maximum soil concentrations for cadmium, copper, lead and zinc exceeding the applicable soil quality guideline. Exceedances of the applicable guidelines and background data occurred in 1% (1 of 103) of cadmium samples; 3% (3 of 103) of copper samples; 28% (32 of 114) lead samples; and 28% (29 of 103) of zinc samples (**Table 5-8**). Maximum concentrations for cadmium, copper, lead, and zinc exceeded their applicable guidelines at 1.5-fold, 1.3-fold, 50-fold, and 8-fold, respectively. Strontium was carried forward as an initial COPC as it is expected as a result of previous site activities; however, no strontium vegetation/soil invertebrate health-based guideline was identified and as such, potential risk to vegetation related to strontium in soil is an uncertainty. Based on these considerations, lead and zinc were carried forward for further assessment for terrestrial vegetation and soil invertebrates. The locations of the exceedances of the vegetation and soil invertebrate health-based guidelines are provided in **Figure 10, Appendix A**.

For mammalian and avian ROC, the 90th percentile concentration exceeded the applicable guidelines for cadmium, lead, strontium (ground feeding mammals, but not top consumer mammals), and vanadium. The exceedances of individual soil samples over the applicable guidelines are provided in **Table J-1, Appendix J**.

Cadmium, lead, strontium, and vanadium were carried forward for further assessment for mammalian and avian ROC. The locations of the exceedances of the mammalian and avian health-based guidelines or all metals other than lead are provided in **Figure 11, Appendix A**, while the lead soil sample exceedances for birds and mammals are provided in **Figure 12, Appendix A**.

Based on the COPC identification steps and considerations described above, a number of COCs in soil were carried forward for assessment as listed below:

COPCs for Terrestrial Vegetation and Soil Invertebrates:

- Lead; and
- Zinc.

COPCs for Small Mammals and Birds:

- Cadmium;
- Lead;
- Strontium; and
- Vanadium.

5.2.8.2

COPCs in Site Surface Water

Surface water samples from the settling/tailings ponds and wetland areas within the study boundary, as well as off-site in Lake Enon were compared to freshwater aquatic life guidelines. In addition, a surface water sample was collected during a precipitation event from a discharge pipe at the edge of the site that discharges into Lake Enon (SW-D). This sample was also compared to the freshwater aquatic life guidelines. The specific source of the guideline used in the COPC screening are identified in the ecological health screening tables (**Tables J-2 and J-3, Appendix J**), while the original guideline sources are provided below.

Canadian Council of Ministers of the Environment (CCME)

The Canadian Water Quality Guidelines (CWQG) for the Protection of Freshwater Aquatic Life are intended to protect all forms of aquatic life and all aspects of aquatic life cycles, including the most sensitive life stage of the most sensitive species over the long term. Details regarding surface water guideline bases and derivation are provided in their respective fact sheets available from CCME (<https://ccme.ca/en/summary-table>).

Federal Environmental Quality Guidelines (ECCC)

Federal Environmental Quality Guidelines (FEQGs) have been developed by Environment and Climate Change Canada under the federal Chemicals Management Plan to support federal environmental quality monitoring, risk assessment, and risk management activities on substances for which CCME guidelines do not yet exist or are not reasonably expected to be updated in the near future (Government of Canada, 2023). Details on surface water guideline bases and derivation are provided on the Government of Canada's website (<https://www.canada.ca/en/health-canada/services/chemical-substances/fact-sheets/federal-environmental-quality-guidelines.html>).

Michigan Department of Environment, Great Lakes and Energy

Aquatic life, final chronic values (reported as MDEQ in NS PSS (2021)). Searched October 2023. <https://www.michigan.gov/egle/about/organization/water-resources/assessment-michigan-waters/rule-57-water-quality-values>.

British Columbia Contaminated Sites Regulation (BC CSR)

The British Columbia Contaminated Sites Regulations (BC CSR) Schedule 3.2 - Generic Numerical Water Standards for Aquatic Life (February 2023) were applied in the screening where other guidelines were not available. Guidelines are available at https://www.bclaws.gov.bc.ca/civix/document/id/complete/statreg/375_96_08.

Results of Ecological Health-Based Surface Water Screening

The following surface water analytical data from the study area were screened to identify COPCs for freshwater aquatic life:

- On-site settling/tailings ponds samples and the discharge sample to Lake Enon (collected up to 2023);
- Lake Enon samples (collected up to 2024);
- On-site wetland surface water samples (2023 and 2024); and
- Watercourses samples (collected in 2024 only).

Study boundary surface water screening results are provided in **Table 5-9** and discussed in the following sections. Details of the ecological screening for metals in site surface water are provided in **Tables J-2 and J-3, Appendix J**. Bismuth, tin, BTEX, PHCs, VOCs, and PAHs were not detected in any surface water samples with the exception of phenanthrene in one sample, which was reported at the detection limit of 0.01 µg/L (and considered essentially equivalent to the detection limit; see **Table J-2, Appendix J**).

Where available, the applicable guidelines were greater than the detection limits. As such, bismuth, tin, BTEX, PHCs, VOCs and PAHs were not carried forward for additional assessment.

Surface water concentrations from the settling/tailings ponds, Lake Enon, the discharge pipe to Lake Enon, and the wetland areas were compared to the generic freshwater aquatic life surface water guidelines (**Tables J-2 and J-6, Appendix J**). The exception to this was aluminium, which was compared

to an adjusted CCME guideline of 100 µg/L since all surface water samples had a pH >6.5. The CCME provides equations to adjust freshwater aquatic life guidelines for some metals using various site-specific water quality parameters (e.g., pH, hardness, DOC). Where metals exceeded the generic surface water guidelines in **Tables J-2 and J-6, Appendix J**, adjusted guidelines were calculated. Where exceedances were noted, and the guideline could be modified with site-specific water quality parameters, the guidelines were adjusted to incorporate the site-specific data (**Tables J-3 and J-7, Appendix J**). **Table 5-9** provides the exceedances of surface water exceedances of freshwater aquatic life guidelines that have been adjusted where available.

No guideline was available for titanium; however, it was detected in 9 of 31 samples up to a maximum of two times the detection limit (**Tables J-2 and J-6, Appendix J**). Titanium was not carried forward for further assessment given its low frequency of detection and given that it has not been identified as a site-related chemical.

Table 5-9: Ecological Screening of Surface Water for the Lake Enon Former Mill Site

| Chemical | Ecological Surface Water Quality Benchmark (SoQGE) (mg/L) | Maximum Background Concentration (mg/L) | Maximum Measured Concentration (mg/L)* | Exceeds Guideline and Background? (Y/N) |
|--|---|---|--|---|
| Lake Enon, Settling Ponds, and Tailings Ponds Samples | | | | |
| Aluminium | 0.1 ^{a, d} | 0.052 | 0.19 | Y |
| Antimony | 0.009 ^e | <0.0010 | <0.0010 | N |
| Arsenic | 0.005 ^a | <0.0010 | <0.0010 | N |
| Barium | 1 ^c | 0.082 | 0.43 | N |
| Beryllium | 0.00015 ^c | <0.00010 | <0.00010 | N |
| Boron | 1.5 ^a | <0.05 | <0.05 | N |
| Cadmium | 0.00009 ^{a, f} | 0.000015 | 0.00012 | Y |
| Chromium (Total, III+VI) | 0.0089 ^a | <0.0010 | <0.0010 | N |
| Cobalt | 0.001 ^b | <0.00040 | <0.00040 | N |
| Copper | 0.002 ^a | 0.00051 | 0.0013 | N |
| Iron | 0.3 ^a | <0.05 | 0.16 | N |
| Lead | 0.001 ^a | 0.0018 | 0.0071 | Y |
| Manganese | 0.43 ^a | 0.017 | 0.14 | N |
| Mercury | 0.000026 ^a | <0.000013 | <0.000013 | N |
| Molybdenum | 0.073 ^a | <0.0020 | <0.0020 | N |
| Nickel | 0.025 ^a | <0.0020 | <0.0020 | N |
| Selenium | 0.001 ^a | <0.00050 | 0.00085 | N |
| Silver | 0.00025 ^a | <0.00010 | <0.00010 | N |
| Strontium | 21 ^c | 1.8 | 16 | N |

| Chemical | Ecological Surface Water Quality Benchmark (SoQG _E) (mg/L) | Maximum Background Concentration (mg/L) | Maximum Measured Concentration (mg/L)* | Exceeds Guideline and Background? (Y/N) |
|----------------------------|--|---|--|---|
| Thallium | 0.0008 ^a | <0.00010 | <0.00010 | N |
| Titanium | NGA | <0.0020 | 0.004 | NA |
| Uranium | 0.015 ^a | <0.00010 | 0.0015 | N |
| Vanadium | 0.12 ^b | <0.0020 | <0.0020 | N |
| Zinc | 0.007 ^a | 0.0056 | 0.028 | Y |
| Watercourse Samples | | | | |
| Aluminium | 0.1 ^{a, d} | - | 0.18 | Y |
| Antimony | 0.009 ^e | - | <0.001 | N |
| Arsenic | 0.005 ^a | - | <0.001 | N |
| Barium | 1 ^c | - | 0.22 | N |
| Beryllium | 0.00015 ^c | - | <0.0001 | N |
| Boron | 1.5 ^a | - | <0.05 | N |
| Cadmium | 0.00009 ^{a, f} | - | 0.00011 | Y |
| Chromium (Total, III+VI) | 0.0089 ^a | - | <0.001 | N |
| Cobalt | 0.001 ^b | - | <0.0004 | N |
| Copper | 0.002 ^a | - | 0.00075 | N |
| Iron | 0.3 ^a | - | 0.18 | N |
| Lead | 0.001 ^a | - | 0.0055 | Y |
| Manganese | 0.43 ^a | - | 0.088 | N |
| Mercury | 0.000026 ^a | - | - | NA |
| Molybdenum | 0.073 ^a | - | <0.002 | N |
| Nickel | 0.025 ^a | - | <0.002 | N |
| Selenium | 0.001 ^a | - | 0.00074 | N |
| Silver | 0.00025 ^a | - | <0.0001 | N |
| Strontium | 21 ^c | - | 7.9 | N |
| Thallium | 0.0008 ^a | - | <0.0001 | N |
| Titanium | NGA | - | 0.0049 | NA |
| Uranium | 0.015 ^a | - | 0.00074 | N |
| Vanadium | 0.12 ^b | - | <0.002 | N |
| Zinc | 0.007 ^a | - | 0.036 | Y |
| Wetland Samples | | | | |
| Aluminium | 0.1 ^{a, d} | - | 0.13 | Y |

| Chemical | Ecological Surface Water Quality Benchmark (SoQG _E) (mg/L) | Maximum Background Concentration (mg/L) | Maximum Measured Concentration (mg/L)* | Exceeds Guideline and Background? (Y/N) |
|------------|--|---|--|---|
| Barium | 1 ^c | - | 0.89 | N |
| Cadmium | 0.00009 ^a | - | 0.00012 | Y |
| Cobalt | 0.001 ^b | - | 0.00097 | N |
| Copper | 0.002 ^a | - | 0.00099 | N |
| Iron | 0.3 ^a | - | 5.5 | Y |
| Lead | 0.001 ^a | - | 0.033 | Y |
| Manganese | 0.43 ^a | - | 7.6 | Y |
| Molybdenum | 0.073 ^a | - | 0.0026 | N |
| Selenium | 0.001 ^a | - | 0.0011 | Y |
| Strontium | 36 ^c | - | 20 | N |
| Titanium | NGA | - | 0.0034 | N |
| Uranium | 0.015 ^a | - | 0.0014 | N |
| Zinc | 0.007 ^a | - | 0.052 | Y |

Notes:

NGA = no guideline available; NA = not applicable; Y = yes; N = no.

N = 22 (Lake Enon and settling/tailings ponds); N = 4 (watercourse areas); N = 9 (wetland areas)

Bold values indicate exceedances over the guideline and background; or if no guideline was available, bold indicates the maximum site concentration exceeded the guideline.

Italicized values indicate exceedances over the guideline, but not background.

- No background surface water concentrations in area wetlands were available, and as such, background is noted as “-” in the table above.
- Where applicable, guidelines taken from Nova Scotia Tier II Pathway-Specific Standards for Surface Water - Freshwater, Table 5. The original source is indicated below.
- Maximum concentrations were determined using all data from Lake Enon, settling ponds, tailings ponds and wetland data prior to 2024 as well as watercourse and wetland data from 2024.
- CCME Freshwater Aquatic Life Guidelines; <https://ccme.ca/en/summary-table>.
- FEQG (Cited In: NS PSS, 2021 Tier II pathway specific standards for ecological surface water; freshwater aquatic life).

Freshwater Aquatic Life

For samples collected from the settling/tailings ponds, the discharge to Lake Enon sample, and Lake Enon, exceedances were noted for aluminium, cadmium, lead and zinc (**Tables J-2 and J-6, Appendix J**). The CCME provides equations to adjust the freshwater aquatic life guidelines for aluminium, cadmium, lead, and zinc. When guidelines for these metals were adjusted, exceedances were noted for aluminium, lead, and zinc (**Tables J-3 and J-7, Appendix J**). **Figure 13A and 13B, Appendix A** shows the location of exceedances of the adjusted guidelines for the surface water samples.

Results of the screening of the adjusted guidelines in **Table J-3, Appendix J** show that there were no guideline exceedances in the Lake Enon surface water samples. Two samples from the settling/tailings ponds exceeded the pH-adjusted lead guideline (i.e., SW4 and SW5); however, the location where these samples were collected was not identified as fish habitat in the aquatic habitat assessment (**Section 3.3**). Background concentrations also exceeded the lead guideline, but to a lesser degree.

The discharge to Lake Enon sample (SW-D) also exceeded the adjusted lead guideline, in addition to the adjusted aluminium and adjusted zinc guidelines (**Table J-3, Appendix J**). Similar to the SW4 and SW5 samples, the discharge sample is not considered to represent freshwater aquatic life habitat. Total suspended solids were not detected in the sample collected at SW-D during the rainfall event. Based on the two samples collected from this discharge point, the rate at which sediments/fines may be flowing from the site into Lake Enon does not appear to be significantly increased during a routine heavy rainfall event (**Appendix C**). This is reinforced by the results of the hydrologic and sediment transport study (Dillon, 2024) which noted that in a typical year, total suspended solid (TSS) contributions to contaminated sediment being transported off-site to Lake Enon would be considered de minimis. While in a 1:100-year scenario, the model simulated suspended solids entering Lake Enon over a range of particle sizes.

Sediment and surface water samples were collected in each of the on-site watercourses that had potential fish habitat (i.e., WC1, WC2 and WC3). One sample of each media type was collected in WC2 and WC3 while a parent and a field duplicate were collected in WC1. Each of the watercourse sample locations exceeded the adjusted guideline for aluminium, lead (field duplicate of WC1 only) and zinc (**Table J-7, Appendix J**). Lead exceedances were reported at concentrations up to 2-fold above the guideline, zinc exceedances by up to 2.9-fold, and aluminium exceedances by up to 1.8-fold. The sample collected from WC1 (plus its field duplicate) exceeded the guideline for cadmium by 1.3-fold. In addition, one wetland sample exceeded the adjusted guideline for lead in the wetland (24-WL3-SW4).

Aluminum is known to be essential for aquatic life. The maximum aluminium concentration in site soils was less than the maximum background soil concentration (**Section 5.2.8.1**), as such, aluminium concentrations in on-site surface waters are unlikely to be site-related. Given this, and that aluminium is an essential element for aquatic life, aluminum was not carried forward for further assessment. Given the limited number of samples collected, the remaining chemicals having exceedances over the adjusted

guidelines in surface water from the watercourses were carried forward as COPCs (i.e., cadmium, lead, and zinc).

Based on the results noted above regarding surface water within the study boundary and off-site in Lake Enon, no COPCs in surface water were carried forward for freshwater aquatic life in Lake Enon, while cadmium, lead and zinc were carried forward as COPCs in the watercourses based on the available data.

Final COPCs Surface Water (Freshwater Aquatic Life)

Lake Enon

- None.

On-site Watercourses

- Cadmium;
- Lead; and
- Zinc.

Amphibians

Amphibians are considered to be one of the two most endangered groups of vertebrates (Ortiz-Santaliestra et al., 2018); however, they are generally not included in ecological risk assessments due to limited toxicity data and few technical guidance documents. There is still a paucity of toxicity data on the terrestrial life stages of amphibians; however, published comprehensive reviews of amphibian ecotoxicology are available (e.g., Sparling et al. 2000, 2010). In Canada, an amphibian risk assessment guidance has been recently developed through the FCSAP program (FCSAP, 2019).

Some locations within the study boundary provide suitable habitat for amphibians, particularly within the wetland areas that were identified as part of the wetland assessment (**Section 3.2, Appendix D**). Both pickerel frog and wood frog were noted on the site during the wetland habitat assessment (**Appendix D**).

The location of the four surface water and sediment samples that were collected from within the study boundary wetland areas in 2023 and analyzed for metals are reported in **Figure 13, Appendix A** and **Figure 4, Appendix D**. The wetland surface water and sediment analytical results for metals can be found in **Appendix E**. An additional five surface water samples and six sediment samples were collected within the wetland areas and analyzed for metals in 2024 and are reported in **Tables J-3 to J-6, Appendix J** and **Figure L3, Appendix L**.

There are no surface water, sediment, or soil health-based benchmarks specifically derived for amphibians. While freshwater and sediment guidelines were not necessarily derived to be protective of amphibians, some amphibian toxicity data are available for metals, and some of the more recent guidelines do incorporate amphibian data into their derivation (e.g., CCME freshwater quality guideline for cadmium). FCSAP (2019) examined water quality guidelines for several metals (i.e., lead, mercury,

cadmium, and zinc) from various jurisdictions and noted that most water quality guidelines appear to provide adequate protection of amphibians for the metals evaluated. As such, freshwater aquatic life benchmarks were used for amphibians in addition to relevant amphibian toxicity data. The comparison of wetland surface water data to freshwater aquatic life guidelines is provided in **Tables J-2 and J-6 (Appendix J)** and summarized in **Table 5-9**. No background wetland surface water data were available for comparison purposes.

The Canadian Water Quality Guidelines (CWQG) for the Protection of Freshwater Aquatic Life are intended to protect all forms of aquatic life and all aspects of aquatic life cycles, including the most sensitive life stage of the most sensitive species over the long term. Details on surface water guideline bases and derivation are provided in their respective fact sheets available from CCME (<https://ccme.ca/en/summary-table>).

Based on the screening of the wetland surface water concentrations to freshwater aquatic life benchmarks, exceedances were noted for aluminium (1 of 9 samples; Wetland #1-B only), cadmium (1 of 9 samples; Wetland #1-B only), iron (7 of 9 samples; Wetland #1-B; Wetland #2; Wetland #3; 24-WL3-SW1; 24-WL3-SW3; 24-WL3-SW4; 24-WL3-DUPA), lead (7 of 9 samples; Wetland #1-B; Wetland #2; Wetland #3; 24-WL3-SW2; 24-WL3-SW3; 24-WL3-SW4; 24-WL3-DUPA), manganese (6 of 9 samples; Wetland #2; Wetland #3; 24-WL3-SW1; 24-WL3-SW3; 24-WL3-SW4; 24-WL3-DUPA), selenium (1 of 4 samples; Wetland #1-B only), 7 of 9 samples for zinc (see **Table 5-9** above and **Table J-2, Appendix J**). While iron and manganese were exceeded in surface water, as discussed previously for human health in **Section 4.2.4.1**, these parameters are related more to background conditions than site contamination. As such, iron and manganese were not carried forward for further assessment.

While no pH data were available for the wetland surface water samples, the aluminium freshwater aquatic life surface water quality guideline was adjusted for the wetland samples assuming the pH of the wetlands were > 6.5. All surface water samples (in the settling/tailings ponds, Lake Enon, and in the watercourses) had a pH > 6.5. The surface water samples nearest to the wetland samples (SW6 and SW7) had a pH > 8. Wetland sample Wetland #1-B (0.13 mg/L) exceeded the freshwater aquatic life guideline of 0.1 mg/L by 1.3-fold. This slight exceedance of only one of the 4 wetlands samples would not likely drive any potential risks to amphibians within the study boundary. In addition, soil from within the study boundary were all within the background range and as such, elevated aluminium is most likely natural and not site related.

The following chemicals were carried forward as COPCs in surface water for amphibians and discussed further in the amphibian risk characterization (**Section 5.6.4**):

- Cadmium;
- Lead;
- Selenium; and
- Zinc.

COPCs in Site Sediment

Sediment samples were assessed separately for the settling/tailings ponds and Lake Enon, and the wetland areas sampled within the study boundary. Sediment screening results are provided in **Table 5-10** and discussed in the following sections. Details of the ecological screening for site sediments are provided in **Tables J-4 and J-8, Appendix J**, while the specific guideline sources are provided below.

Canadian Council of Ministers of the Environment (CCME)

The applicable guidelines for freshwater sediments are the CCME Canadian Sediment Quality Guidelines for the Protection of Aquatic Life (<http://ceqg-rcqe.ccme.ca/en/index.html>). CCME has two types of sediment guidelines: Interim Sediment Quality Guidelines (ISQGs) and Probable Effects Levels (PELs). The ISQGs are essentially threshold effect levels (TELS), which represent the concentration of a contaminant below which adverse effects are not anticipated (CCME, 1995). A PEL is defined by CCME (1995) as the concentration of a contaminant above which adverse biological effects are usually or always observed. As noted in **Section 5.2.8**, the PEL guidelines were preferentially applied for the COPC screening.

British Columbia Ministry of Environment and Climate Change Strategy Approved and Working Sediment Quality Guidelines

The BC MECCS (2021; 2023) Working and Approved Sediment Quality Guidelines (freshwater). These guidelines represent safe levels of substances that protect different water uses including freshwater aquatic life. The BC MECCS (2023) notes that exceeding the guideline does not imply that unacceptable risks exist, but rather that the potential for adverse effects may be increased and additional investigation may be required.

Australian and New Zealand Guidelines for Fresh and Marine Water Quality (ANZECC/ARMCANZ; Simpson et al., 2013)

The Australian and New Zealand Guidelines for Fresh and Marine Water Quality (ANZECC/ARMCANZ) provide comprehensive guidance on assessing and managing the quality of freshwater and marine environments in Australia and New Zealand. These guidelines offer a framework for water quality monitoring and management, covering aspects such as water quality objectives, environmental indicators, and assessment methodologies to protect and sustainably manage aquatic ecosystems and water resources in the region. They are an essential reference for policymakers, researchers, and environmental practitioners involved in water quality management in Australia and New Zealand. The derivation of these values can be found at <https://publications.csiro.au/rpr/pub?pid=legacy:965>.

Results of Ecological Health-Based Sediment Screening

Sediment analytical results was screened to identify COPCs for both freshwater aquatic life (on-site settling/tailings ponds, and Lake Enon) and for amphibians (wetland samples). COPC identification for each receptor group is discussed below. One sediment sample collected in Lake Enon to the far north of

the site was used for background comparison for the Lake Enon samples collected in the vicinity of the site. No background wetland sediment data were available for comparison purposes.

Site sediment screening results are provided in **Table 5-10** and discussed in the following sections. Details of the ecological screening for site sediment are provided in **Table J-4, Appendix J. Figure 14A and 14B (Appendix A)** shows the locations of guideline exceedances for metals in sediment. Results of the sediment screening were used to identify COCs for freshwater aquatic life and amphibians.

Chemicals that were not detected in any of the analyzed sediment samples and were not associated with current and historical activities at the site were excluded from further assessment. For site sediments, the following analytes were not detected in sediment samples and were excluded from further assessment: antimony, bismuth, boron, cyanide (free and total), all PAHs, benzene, toluene, ethylbenzene, xylenes, and all VOCs (**Appendix E**). For site wetland sediment, the following analytes were not detected in sediment samples and were excluded from further assessment: antimony, beryllium, bismuth, boron, mercury, and tin. Analytical results for these parameters are provided in **Appendix J**.

For completeness, the RDL for parameters with concentrations < the RDL were cross-referenced with the appropriate ecological-health based guideline, and all RDLs were below the applicable guideline; therefore, none of the non-detected parameters were carried forward in the ERA.

No guidelines were available for aluminium, barium, beryllium, cobalt, lithium, molybdenum, rubidium, strontium, thallium, tin, uranium, and vanadium (**Table 5-10**). Many of the metals with reported concentrations greater than the laboratory RDLs and no regulatory benchmarks are well established as being major or common naturally occurring elements in all freshwater bodies and tend to have a high natural abundance in sediment that is rarely related to former or current activities at a given site. Given the known former activities within the study boundary, the presence of aluminium, barium, cobalt, lithium, molybdenum, rubidium, thallium, tin, uranium, and vanadium, cannot be clearly attributed to former or current site activities. Although, concentrations of these metals in some of the site samples are generally greater than in the background sediment. Strontium was considered to be a site-related chemical. Some of the metals listed in **Table 5-10** (e.g., aluminium, iron) are known essential nutrients for all aquatic life. As such, the absorption, metabolism, distribution and elimination of these elements are physiologically or biochemically regulated such that adverse effects would not be expected to occur except in conditions of extremely high exposure. Ambient environmental exposure to these elements in sediment, even when/if present at elevated concentrations, does not constitute a situation of extreme exposure.

Table 5-10: Ecological Screening of Sediment for the Lake Enon Former Mill Site

| Chemical | Ecological Sediment Quality Benchmark (SoQGE) (mg/kg) | Maximum Background Concentration (mg/kg) | Maximum Measured Concentration (mg/kg)* | Exceeds Guideline and Background? (Y/N) |
|--|---|--|---|---|
| Lake Enon, Settling Ponds, and Tailings Ponds Samples | | | | |
| Aluminium | NGA | 7,800 | 30,000 | NA |
| Arsenic | 17.0 ^a | 3.5 | 13 | N |
| Barium | NGA | 230 | 2,100 | NA |
| Beryllium | NGA | <1.0 | 1.5 | NA |
| Cadmium | 3.5 ^a | 1.2 | 23 | Y |
| Chromium (Total, III+VI) | 90 ^a | 9.3 | 29 | N |
| Cobalt | NGA | 5.2 | 79 | NA |
| Copper | 197 ^a | 7.2 | 350 | Y |
| Iron | 43,766 ^b | 13,000 | 34,000 | N |
| Lead | 91.3 ^a | 71 | 5,700 | Y |
| Lithium | NGA | 11 | 40 | NA |
| Manganese | 1100 ^b | 2,100 | 23,000 | Y |
| Mercury | 0.486 ^a | <0.10 | 0.3 | N |
| Molybdenum | NGA | <2.0 | 9.5 | NA |
| Nickel | 75 ^b | 9.1 | 51 | N |
| Rubidium | NGA | 6.3 | 20 | NA |
| Selenium | 2.0 ^c | <0.50 | 8.7 | Y |
| Silver | 0.5 ^b | <0.50 | 2.8 | Y |
| Strontium | NGA | 150 | 17,000 | NA |
| Thallium | NGA | 0.11 | 0.82 | NA |
| Tin | NGA | <1.0 | 1.9 | NA |
| Uranium | NGA | 1.2 | 8.2 | NA |
| Vanadium | NGA | 16 | 43 | NA |
| Zinc | 315 ^a | 180 | 3,100 | Y |
| PHC F1-BTEX (C6-C10-BTEX) | NGA | - | ND | NA |
| EPH >C10-C16 | NGA | - | 86 | NA |
| EPH >C16-C21 | NGA | - | 330 | NA |
| EPH >C21-C32 | NGA | - | 490 | NA |
| Modified TPH (Tier 1) | 43 ^{e.g} | - | 910 | Y |
| Watercourse Samples | | | | |
| Aluminium | NGA | - | 9,600 | NA |
| Arsenic | 17.0 ^a | - | 21 | Y |
| Barium | NGA | - | 620 | NA |

| Chemical | Ecological Sediment Quality Benchmark (SoQGE) (mg/kg) | Maximum Background Concentration (mg/kg) | Maximum Measured Concentration (mg/kg)* | Exceeds Guideline and Background? (Y/N) |
|--------------------------|--|---|--|--|
| Cadmium | 3.5 a | - | 4.4 | Y |
| Chromium (Total, III+VI) | 90 a | - | 14 | N |
| Cobalt | NGA | - | 11 | NA |
| Copper | 197 a | - | 38 | N |
| Iron | 43,766 b | - | 20,000 | N |
| Lead | 91.3 a | - | 270 | Y |
| Lithium | NGA | - | 18 | NA |
| Manganese | 1,100 b | - | 3,000 | Y |
| Molybdenum | NGA | - | 2 | NA |
| Nickel | 75 b | - | 21 | N |
| Rubidium | NGA | - | 7.8 | NA |
| Selenium | 2.0 c | - | 1.1 | N |
| Silver | 0.5 b | - | 1.6 | Y |
| Strontium | NGA | - | 3,000 | NA |
| Thallium | NGA | - | 0.38 | NA |
| Uranium | NGA | - | 2.1 | NA |
| Vanadium | NGA | - | 28 | NA |
| Zinc | 315 a | - | 840 | Y |
| Wetland Samples | | | | |
| Aluminium | NGA | - | 13,000 | NA |
| Arsenic | 17.0 a | - | 15 | N |
| Barium | NGA | - | 1,200 | NA |
| Cadmium | 3.5 a | - | 3.8 | Y |
| Chromium (Total, III+VI) | 90 a | - | 29 | NA |
| Cobalt | NGA | - | 11 | NA |
| Copper | 197 a | - | 34 | N |
| Iron | 43766 b | - | 23,000 | N |
| Lead | 91.3 a | - | 14,000 | Y |
| Lithium | NGA | - | 23 | NA |
| Manganese | 1100 b | - | 3,200 | Y |
| Molybdenum | NGA | - | 2.1 | NA |
| Nickel | 75 b | - | 49 | N |
| Rubidium | NGA | - | 13 | NA |

| Chemical | Ecological Sediment Quality Benchmark (SoQGE) (mg/kg) | Maximum Background Concentration (mg/kg) | Maximum Measured Concentration (mg/kg)* | Exceeds Guideline and Background? (Y/N) |
|-----------|---|--|---|---|
| Selenium | 2.0 ^c | - | 0.95 | N |
| Silver | 0.5 ^b | - | 2.2 | Y |
| Strontium | NGA | - | 5,400 | NA |
| Thallium | NGA | - | 0.44 | NA |
| Uranium | NGA | - | 3.1 | NA |
| Vanadium | NGA | - | 30 | NA |
| Zinc | 315 ^a | - | 770 | Y |

Notes:

N = 25

NGA = no guideline available; NA = not applicable; Y = yes; N = no' ND = not detected.

Bold values indicate exceedances over the guideline and background.*Italicized* values indicate exceedances over the guideline, but not background.

No background surface water concentrations in area wetlands were available, and as such, background is noted as “-” in the table above.

Where applicable, guidelines taken from Nova Scotia Tier II Pathway-Specific Standards for Sediment - Freshwater, Table 4. The original source is indicated below.

No amphibian sediment guidelines were available; therefore, the CCME sediment freshwater aquatic life guidelines were used in the screening of this receptor. These guidelines are generally protective of amphibians as discussed in **Section 5.2.8.2**.

*Maximum concentrations were determined using all data from Lake Enon, settling ponds, tailings ponds and wetland data prior to 2024 as well as watercourse and wetland data from 2024.

- a. CCME Canadian Sediment Quality Guidelines for the protection of freshwater aquatic life; Probable Effect Levels (PEL).
- b. BC MECCS (2021) British Columbia Working Sediment Quality Guidelines. Upper SWQG for freshwater aquatic life - Table 2.
- c. BC MECCS (2023) British Columbia Approved Water Quality Guidelines for Aquatic Life (selenium, Table 36)
- d. Simpson et al. 2013. Revision of the ANZECC/ARMCANZ Sediment Quality Guidelines.
<https://doi.org/10.4225/08/5894c6184320c>
- e. Atlantic RBCA, 2021
- f. Guideline is for benzo(b+j+k)fluoranthene
- g. Guideline is for modified TPH (lube) as all samples had lube oil resemblance.
- h. US EPA, 2008

Freshwater Aquatic Life

Maximum concentrations of cadmium, copper, lead, manganese, selenium, silver, zinc (in the settling/tailings ponds and in Lake Enon), and modified TPH (in the settling/tailings ponds), exceeded their respective ecological-health based sediment quality guideline. Multiple exceedances of sediment guidelines were noted for cadmium (31/43), lead (34/43), manganese (38/43), selenium (7/43), silver (36/43) and zinc (36/43) (**Table 5-10; Tables J-4 and J-8, Appendix J**). **Figure 14A and 14B, Appendix A** shows the location of samples that have at least one sediment exceedance. These chemicals were carried forward for further discussion with the exception of copper and manganese. Copper was only

exceeded in 1/43 of the sediment samples collected by <2-fold and also does not seem to be site related. Similarly, manganese has been previously noted to be likely related to local background conditions. Given these consideration, copper and manganese were not carried forward for additional assessment. Strontium in sediments was considered to be a site-related chemical and carried forward as a COPC for further discussion.

Final COPCs Sediments (Freshwater Aquatic Life)

- Cadmium;
- Lead;
- Selenium,
- Silver,
- Strontium;
- Zinc; and
- Modified TPH.

The potential risk to freshwater aquatic as a result of potential exposures to these COPCs is discussed in **Section 5.5.1.3**.

Amphibians

Based on the screening of wetland sediment samples, exceedances were noted for cadmium in 1 of 10 samples (concentration 3.8 mg/kg; guideline of 3.5 mg/kg), lead in 10 of 10 samples (concentrations of 100, 280, 120, 6,100, 7,100, 9,700, 10,000 and two samples at 13,000 mg/kg; guideline of 91.3 mg/kg), manganese in 10 of 10 samples (concentrations of 1700, 3200, 1900, 2,200, 2,400, 2600, 3,000 and 3,100 mg/kg; guideline of 1,100 mg/kg), silver in 8 of 10 samples (concentrations of 0.92, 1.0, 1.2, 1.4, 1.5, 1.9 and 2.2 mg/kg; guideline of 0.5 mg/kg), and zinc in 5 of 10 samples (concentration of 330, 400, 450 and 770 mg/kg; guideline is 315). While manganese exceeded the freshwater aquatic life sediment guideline, as discussed previously, this parameter is related more to background conditions than site contamination. As such, manganese was not carried forward for further assessment. Strontium in sediments was considered to be a site-related chemical and carried forward as a COPC for further discussion.

The following chemicals were carried forward as COPCs in sediment for amphibians and are discussed further in the amphibian risk characterization section (**Section 5.5.1.4**).

Final COPCs Sediments (Amphibians)

- Cadmium;
- Lead;
- Silver;
- Strontium; and
- Zinc.

COPCs in Site Groundwater

The regulatory ecological health-based groundwater quality benchmarks that were used to identify COPCs in site groundwater are briefly described below. The specific source of the guideline used in the COPC groundwater screening are identified in the ecological health screening tables (**Table J-5, Appendix J**).

FCSAP Federal Interim Groundwater Quality Guidelines for Federal Contaminated Sites (FCSAP, 2016)

The applicable assessment standards for groundwater are the FCSAP Federal Interim Groundwater Quality Guidelines for Federal Contaminated Sites Generic Guidelines for Agricultural Land Use. Guidelines protective of the soil organism's direct contact and freshwater life exposure pathways were applied herein.

Ontario Ministry of the Environment Conservation and Parks (MECP, 2011)

The environmental health-based groundwater guidelines were selected from Table 6 – Groundwater Shallow Potable (GW3) for coarse textured soil. Details on the basis and derivation of these benchmarks is provided (MECP, 2011).

Results of Ecological Health-Based Groundwater Screening

Site groundwater screening results are provided in **Table 5-11**, and discussed in the following sections. Details of the ecological screening for site groundwater are provided in **Table J-5, Appendix J. Figure 13A and 13B, Appendix A** shows the location of the groundwater monitoring wells. The full sediment data set can be found in **Appendix E**. Results of the groundwater screening were used to identify COCs for freshwater aquatic life.

Chemicals that were not detected in any of the analyzed groundwater samples and were not associated with current and historical activities at the site were excluded from further assessment. For metals in groundwater, this included beryllium, bismuth, chromium, cyanide, mercury, silver, thallium, tin, titanium, and vanadium (**Table J-5, Appendix J**).

For parameters that were detected, the following analytes did not have an applicable guideline for comparison: calcium, magnesium, manganese, potassium, sodium, and strontium (**Table 5-11**). Calcium, magnesium, sodium, and potassium are essential nutrients for all animal life (NRC, 2005; US EPA, 2018). Their absorption, metabolism, distribution, and elimination physiologically or biochemically regulated such that adverse effects would not be expected to occur except in conditions of extremely high exposure. The US EPA (Region 4 supplemental ERA guidance document; US EPA, 2018), notes that calcium, magnesium, sodium, and potassium can be eliminated from ERAs given their essentiality. Further, they are not associated with any known former or current site activities or structures. The contribution of these chemicals to overall risks to aquatic life within the study boundary or off-site in Lake Enon would be expected to be negligible. As such, these parameters were not carried forward for further assessment.

Although strontium does not have a guideline, this parameter is linked to historical site activities and site groundwater concentrations were higher than background concentrations; therefore, strontium was carried forward for further assessment (Table 5-11).

Table 5-11: Ecological Screening of Groundwater for the Lake Enon Former Mill Site

| Chemical | Ecological Groundwater Quality Benchmark (SoQG _E) (mg/L) | | Maximum Background Concentration (mg/L) | Maximum Measured Concentration (mg/L) | Exceeds Guideline and Background? (Y/N) |
|------------|--|--------------|---|---------------------------------------|---|
| Aluminium | 0.10 | ^a | <0.0050 | 0.031 | N |
| Antimony | 2.0 | ^a | <0.0010 | 0.0017 | N |
| Arsenic | 0.005 | ^a | <0.0010 | 0.0022 | N |
| Barium | 2.9 | ^a | 0.4 | 0.26 | N |
| Boron | 36000 | ^b | <0.05 | 0.081 | N |
| Cadmium | 0.000017 | ^a | 0.00035 | 0.00093 | Y |
| Calcium | NGA | | 28 | 340 | NA |
| Cobalt | 52 | ^b | 0.00072 | 0.0053 | N |
| Copper | 0.002 | ^a | <0.00050 | 0.035 | Y |
| Iron | 0.3 | ^a | <0.05 | 0.21 | N |
| Lead | 0.001 | ^a | <0.00050 | 0.0048 | Y |
| Magnesium | NGA | | 2.2 | 64 | N |
| Manganese | NGA | | 0.4 | 3.2 | NA |
| Molybdenum | 0.073 | ^a | 0.007 | 0.026 | N |
| Nickel | 0.025 | ^a | <0.0020 | 0.046 | Y |
| Potassium | NGA | | 3 | 14 | NA |
| Selenium | 0.001 | ^a | <0.00050 | 0.01 | Y |
| Sodium | NGA | | 14 | 14 | N |
| Strontium | NGA | | 0.31 | 39 | NA |
| Uranium | 0.015 | ^a | 0.00069 | 0.012 | N |
| Zinc | 0.03 | ^a | <0.0050 | 0.2 | Y |

Notes:

N = 11

NGA = no guideline available; NA = not applicable; Y = yes; N = no

Bold values indicate exceedances over the guideline and background.

Italicized values indicate exceedances over the guideline, but not background.

^{a.} FCSAP Federal Interim Groundwater Quality Guidelines for Federal Contaminated Sites (FCSAP, 2016). Guidelines for agricultural land use protective of soil organisms direct contact and freshwater aquatic life exposure pathways were applied.

^{b.} Ontario Ministry of the Environment, Conservation, and Parks (MECP, 2011), Table 6 – Groundwater Shallow Potable (GW3) for coarse-textured soil.

Based on the screening of the groundwater concentrations to applicable benchmarks, exceedances were noted for cadmium (11 of 11 samples), copper (2 of 11 samples), lead (4 of 11 samples), nickel (2 of 11 samples), selenium (3 of 11 samples), and zinc (5 of 11 samples); therefore, these COPCs were carried forward for further assessment.

The following chemicals were carried forward as COPCs in groundwater and are further discussed in the risk characterization section (**Section 5.5.1.3**).

Final COPCs Groundwater (Freshwater Aquatic Life)

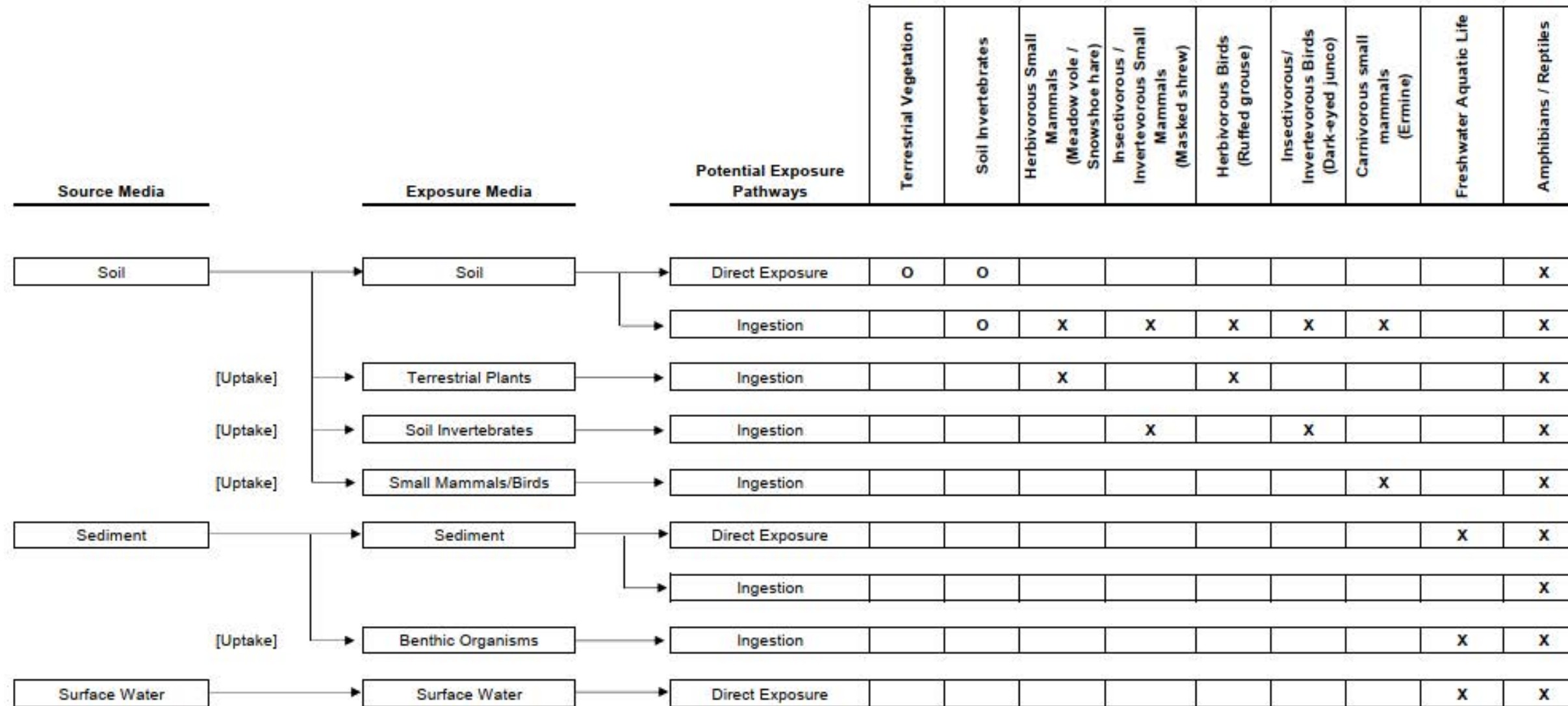
- Cadmium;
- Copper;
- Lead;
- Nickel;
- Selenium;
- Strontium; and
- Zinc.

5.2.9 Conceptual Site Model for Ecological Receptors

A CSM provides a description of relevant exposure pathways for the COPCs that have the potential to contribute to ecological exposures and potential health risks for the selected receptors. The ERA CSM is presented in **Figure 5-2**. The CSM provides a simplified representation of relevant exposure pathways and routes that link the identified COPCs in site environmental media to the selected human receptors.

Based on the COPC screening, COPCs were carried forward for terrestrial vegetation and soil invertebrates, small mammals and birds, freshwater aquatic life, and amphibians. Potential for exposures to the reptilian SAR, the snapping turtle was also carried forward for discussion but no COPCs for this receptor were identified due to a paucity of reptile toxicity data.

Figure 5-2: Conceptual Site Model for Soils for Ecological Receptors at the Lake Enon Former Mill Site



Note:
 1) X - Indicates a potentially complete exposure pathway for quantitative analysis.
 1) O - Indicates an interaction that will be evaluated qualitatively.

5.3 Exposure Assessment

The exposure assessment step of an ERA involves estimating the amount (quantity) of chemicals that are received by ecological receptors. Exposure can be calculated using quantitative approaches (e.g., where exposures for a specific receptor are estimated using models and a variety of receptor input parameters), or can be assessed qualitatively (e.g., where exposures are assumed to equal measured concentrations in environmental media). The latter method 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 an organism.

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

- The concentration of chemicals in various environmental media (e.g., water, soil, 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, soils/dusts);
- The physiological and behavioural characteristics of ecological receptors that determine the actual exposures through interactions with the various pathways (e.g., rates of respiration, water intake, food intake, soils/dusts intake, energy utilization); and
- 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).

The exposure assessment step of an ERA involves estimating the amount (quantity) of chemicals that are received by ecological receptors. Exposure can be calculated using quantitative approaches (e.g., where exposures for a specific receptor are estimated using models and a variety of receptor input parameters), or can be assessed qualitatively (e.g., where exposures are assumed to equal measured concentrations in environmental media).

For the terrestrial vegetation, soil invertebrates, and freshwater aquatic life, potential exposures were assessed qualitatively, and exposures were assumed to equal the measured COPC concentrations in the applicable site media (i.e., soil, sediment, surface water).

Exposures to amphibians were also assessed qualitatively and were assumed to be equal to site surface water and sediment concentrations for their aquatic life stages/exposure pathways. There is a paucity of

toxicity data for land-based exposures for these receptors and as such, the focus was on aquatic-based exposures.

For the remaining ROCs (i.e., vole, shrew, hare, ermine, junco, grouse), COPC exposures were assessed quantitatively using various site-specific assumptions, as well as standard food chain modeling equations, uptake factors (or regression equations), and receptor physiological and behavioural parameters from various regulatory agency ERA guidance documentation and/or the scientific literature. The equations, factors, parameters, and assumptions used in the exposure modelling are described below and in **Appendix K**.

Within the current ERA, the assumed exposure scenario is current site conditions within the study boundary. At this time, it is also assumed that current conditions represent future conditions, as the site is not anticipated to change in the future. The temporal boundaries for the ERA are also current conditions. Spatial boundaries for the ERA exposure scenario are the current areal extent of the study boundary, in addition to considering the potential off-site impacts to Lake Enon. Other properties or other off-site areas are not considered in the ERA.

5.3.1 Equations, Factors, Parameters and Assumptions used in the Quantitative Exposure Modelling

Table 5-12 presents the receptor parameters and assumptions that were used to estimate COPC exposures to the mammalian and avian ROCs that underwent quantitative exposure modelling (i.e., vole, shrew, hare, ermine, junco, and grouse). Equations illustrating how exposures were estimated for the mammalian and avian ROCs in the ERA, via the selected exposure pathways and routes are provided in **Appendix K** along with the ERA modelling results.

Table 5-12: Receptor Parameters and Assumptions for Mammalian and Avian ROCs

| Parameter/ Assumption | Meadow Vole | Common (Masked) Shrew | Snowshoe Hare | Ermine | Ruffed Grouse | Dark-eyed Junco |
|--|---|--|---|--|--|--|
| Body Weight (kg) | 0.0349 (FCSAP, 2012b) | 0.0041 (FCSAP, 2012b) | 1.3 (FCSAP, 2012b) | 0.089 (FCSAP, 2012b) | 0.552 (FCSAP, 2012b) | 0.0188 (Dunning, 1993) |
| Diet Assumptions | 100% terrestrial vegetation (i.e., berries, grains, mushrooms) (FCSAP, 2012b) | 95% soil invertebrates (e.g., insect larvae, slugs, snails, earthworms) and 5% other (assumed terrestrial vegetation) ¹ (FCSAP, 2012b) | 100% terrestrial vegetation (e.g., grasses, sedges) (FCSAP, 2012b) | 100% small mammals (assumed) ¹ | 85% terrestrial vegetation (leaves, berries) and 15% soil invertebrates (e.g., insects and invertebrates) (FCSAP, 2012b) | 60% soil invertebrates and 40% terrestrial vegetation (Dunning, 1993; Reichard and Ketterson, 2012) |
| Average Water Content in Food Items (%) | 85 (Sample and Suter, 1994) | 84 (Sample and Suter, 1994) | 85 (Sample and Suter, 1994) | 67 (Sample and Suter, 1994) | 85 (Sample and Suter, 1994) | 84 (Sample and Suter, 1994) |
| Food Ingestion Rate | 0.33 kg wet food/kg wet BW/day (FCSAP, 2012b) | 0.621 x (body weight (g)) ^{0.564} (U.S. EPA, 1993) | 0.06 kg wet food/kg wet BW/day (FCSAP, 2012b) | 0.11 kg wet food/kg wet BW/day (FCSAP, 2012b) | 0.06 kg dry food/kg wet BW/day (FCSAP, 2012b) | 0.398 x (body weight (g)) ^{0.850} (U.S. EPA, 1993) |
| Soil Ingestion Rate (% of food consumption rate) | 2.4 (Beyer <i>et al.</i> 1994) | 3; based on values reported by U.S. EPA (2005b) using the short- tailed shrew as a surrogate species | 6.3 (FCSAP, 2012b) | 4; based on values reported by U.S. EPA (2005b) using the long-tailed weasel as a surrogate species | 9.3; based on values reported in Beyer <i>et al.</i> (1994) using the wild turkey as a surrogate species | 9.3; based on values reported in Beyer <i>et al.</i> (1994) using the wild turkey as a surrogate species |
| Home Range (ha) | 0.0069 (FCSAP, 2012b) | 0.6 (FCSAP, 2012b) | 1.6 (FCSAP, 2012b) | 1 (FCSAP, 2012b) | 1.0 (FCSAP, 2012b) | 0.833 (Reichard and Ketterson, 2012). |
| Temporal Use Factor (TUF) | 1 (assumed) | 1 (assumed) | 1 (assumed) | 1 (assumed) | 1 (assumed) | 1 (assumed) |
| Residency Factor – R | 1 (assumed) | 1 (assumed) | 1 (assumed) | 1 (assumed) | 1 (assumed) | 1 (assumed) |

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Equations used to estimate how trace metals transfer into plants and soil invertebrates vary by COPC. These uptake equations were sourced from the U.S. EPA Ecological Soil Screening Level (EcoSSL) documents and are presented in **Appendix K** for each COPC and medium. The uptake equations for the COPCs from U.S. EPA (2007b) yield concentrations that are expressed as dry weight. The dry weight values were converted to their equivalent wet weight concentrations using literature-based data for water content of the plant, invertebrate and small mammal prey food items that are consumed by the selected ROCs. Total exposures were estimated for the ROCs, via the selected exposure pathways and routes using the following equations:

Exposure via Terrestrial Vegetation Ingestion

$$EXP_{TV} = \left(\frac{C_P \times FI \times D_F}{BW} \right)$$

Where:

- EXP_{TV} = Exposure from terrestrial vegetation ingestion (mg/kg body weight/day)
- C_P = Calculated terrestrial plant tissue concentration (mg/kg ww)
- FI = Receptor specific food ingestion rate (kg ww/day)
- BW = Receptor specific body weight (kg)
- D_F = Proportion of diet that is terrestrial vegetation (receptor-specific) (%)

Exposure via Soil Invertebrate Ingestion

$$EXP_{SI} = \left(\frac{C_I \times FI \times D_F}{BW} \right)$$

Where:

- EXP_{SI} = Exposure from soil invertebrate ingestion (mg/kg body weight/day)
- C_I = Calculated soil invertebrate tissue concentration (mg/kg ww)
- FI = Receptor specific food ingestion rate (kg ww/day)
- BW = Receptor specific body weight (kg)
- D_F = Proportion of diet that is soil invertebrates (receptor-specific) (%)

Exposure via Small Mammal Ingestion

$$EXP_M = \left(\frac{C_M \times FI \times D_F \times D_o}{BW} \right)$$

Where:

- EXP_M = Exposure from small mammal ingestion (mg/kg body weight/day)
- C_M = Calculated insectivorous small mammal prey tissue concentration (mg/kg ww)
- FI = Food ingestion rate (kg ww/day)
- D_o = Fraction of diet from the subject Site (Assumed 1.0; unitless)
- BW = Receptor specific body weight (kg)
- D_F = Receptor specific diet proportions (%)

Exposure from Incidental Ingestion of Soil

$$EXP_S = \left(\frac{C_S \times SI}{BW} \right)$$

Where:

- EXP_S = Exposure from ingestion of soil (mg/kg body weight/day)
- C_S = Soil exposure point concentration (maximum measured) (mg/kg dw)
- SI = Receptor specific soil ingestion rate (kg/day)
- BW = Receptor specific body weight (kg)

Total Exposure (from soil, terrestrial plant, small mammal, and soil invertebrate ingestion)

$$EXP_{Total} = (EXP_{TV} + EXP_{SI} + EXP_M + EXP_S) \times TUF \times R$$

Where:

- EXP_{Total} = Total exposure (mg/kg body weight/day)
- EXP_{TV} = Exposure from terrestrial vegetation ingestion (mg/kg body weight/day)
- EXP_{SI} = Exposure from soil invertebrate ingestion (mg/kg body weight/day)
- EXP_M = Exposure from small mammal prey ingestion (mg/kg body weight/day)
- EXP_S = Exposure from ingestion of soil (mg/kg body weight/day)
- TUF = Temporal use factor (unitless) (1, assumed)
- R = Residency factor (unitless) (calculated, receptor specific: Area of site/Home range)

5.3.2

Exposure Point Concentrations

The mammalian and avian ROC quantitatively modelled in the ERA, are not sedentary and would be expected to move around the site in search of food and suitable habitat, etc. The focus of the ERA for these receptors is at the population level. Assuming these ecological receptors will spend their entire time and get all their food from locations within the study boundary with the highest concentrations, while highly conservative, is unrealistic. As such, the COPC exposure point concentrations (EPCs) for soils utilized in the modelling equations presented in **Appendix K** were the 95% upper confidence level on the mean (UCML95) soil concentrations calculated with U.S. EPA ProUCL Version 5.2. The same considerations and decision rules previously described in the HHRA (**Section 4.3.2**) applied to the calculation of UCLM95 soil EPCs for ERA purposes. The soil EPCs for each study boundary COPCs are summarized in **Table 5-13**.

Table 5-13: UCLM95-based EPCs for COPCs in Study Boundary Surface Soil

| COPC | Maximum Reported Concentration in Surface Soil (mg/kg) | UCLM95-based EPC (mg/kg) | Comment |
|-------------|---|---------------------------------|--|
| Cadmium | 25 | 3.1 | UCLM 95 95% Approximate Gamma UCL |
| Lead | 30,000 | 4369 | UCLM 95 95% Hall's Bootstrap UCL |
| Strontium | 27,000 | 5917 | UCLM 95 95% Chebyshev (Mean, Sd) UCL |
| Vanadium | 73 | 29 | UCLM 95 95% Student's-t UCL |

Notes:

UCLM95 derived using U.S. EPA ProUCL Version 5.2

5.4**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 taken into an organism or its tissues and the duration of the 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 a duration of exposure, which is necessary to produce a toxic effect in the ROCs (this is referred to as the exposure–response or dose-response relationship). In the effects assessment step of an ERA, information relating to the exposure-response or dose-response relationships of COPCs is evaluated in order to determine an exposure or dose that is acceptable (unlikely to cause harm) in the ROCs selected for evaluation. Such exposures or doses are commonly referred to as toxicity reference values (TRVs). Such values may exist for a number of biological 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 may vary depending on the ROC that is 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 and avian 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 that are in direct contact with an exposure medium (such as freshwater aquatic life in contact with surface water and

sediments). For such receptor groups, the selected TRVs are generally regulatory environmental quality benchmarks that considered these types of organisms in their derivation (e.g., CCME surface water and sediment quality guidelines).

It is important to recognize that when/if ecological TRVs are exceeded by estimated exposures, it does not necessarily imply that there is a risk of adverse ecological effects. Rather, it suggests that further evaluation or consideration of additional lines of evidence may be warranted before reaching final conclusions on the potential for ecological risk. This is discussed further in the risk characterization section below (**Section 5.5**).

For receptors that were not quantitatively modelled (i.e., terrestrial vegetation, soil invertebrates, freshwater aquatic life, and amphibians), the selected TRVs were the applicable benchmarks used in the COPCs screening (**Section 5.2.8**). For amphibians existing toxicity data and other scientific and government sources were consulted to obtain relevant toxicity information. The toxicity data located for amphibian is provided in **Section 5.5.1.4**.

For receptors that were quantitatively modelled (i.e., masked shrew, meadow vole, snowshoe hare, ermine, junco, grouse), published regulatory TRVs were preferentially used where available (i.e., FCSAP, 2021). No published Canadian regulatory TRVs for strontium for birds was identified. As such, the strontium TRV for birds was based on the maximum tolerable level for strontium in poultry (NAS, 2005). The FCSAP (2021) TRVs for cadmium (mammals), lead and vanadium (birds and mammals) are based on no effect concentrations (for reproduction, growth and survival), and were derived to set conservative soil screening values by the US EPA (2005). These NOAEL-based TRVs were selected for use in the ERA modelling. As these TRVs are based on no effect concentrations (generally the lowest no effect concentration of all identified), an exceedance does not indicate a potential risk. Rather it indicates that further examination/discussion is warranted. The selected TRVs and their basis are provided in **Table 5-14**.

Table 5-14: Initial No-Effect Based Toxicity Reference Values (TRVs) for COPCs for Mammalian and Avian ROCs.

| COPC | Avian TRV (Reference) ¹ (mg/kg-BW/d) | Mammalian TRV (Reference) ¹ (mg/kg-BW/d) |
|-------------|--|--|
| Cadmium | 2.1 (CEAEQ, 2012) | 0.77 (US EPA, 2005a) |
| Lead | 1.63 (US EPA, 2005b) | 4.7 (US, EPA, 2005b) |
| Strontium | 147 (NAS, 2005) | 51.3 (CEAEQ, 2012) |
| Vanadium | 0.344 (US EPA, 2005c) | 4.16 (US EPA, 2005c) |

Notes:

1. Federal TRVs selected from the FCSAP (2021) guidance document where available, unless otherwise noted. Original sources upon which the TRV was selected that were provided in FCSAP (2021) are included in the table above.
2. No federal TRV was available and as such, one was selected based on information in NAS, 2005 and Sample et al., 1996). The US EPA TRVs cited above are highly conservative recommended values from the U.S. EPA which are generally based on NOAELs (No-observable-adverse-effect-levels) that are considered to be protective of all birds and all mammals.

The TRVs in **Table 5-14** are generally based on reported NOAELs and LOAELs for reproductive, growth and survival effects from toxicological studies conducted with numerous bird and mammal species. The approach that the U.S. EPA (2007) takes for developing TRVs (i.e., for broad taxonomic classes, such as *Mammalia* and *Aves*) is considered to be a current preferred best practice when feasible, and when sufficient toxicological data exist that are of adequate quality. The approach attempts to address the frequent high variability in NOAEL and LOAEL values that often occur across different toxicological studies with different species, and even across studies conducted using the same species. For example, U.S. EPA (2007) presents information which shows that PAH NOAELs and LOAELs (for reproduction, growth, and survival effects), span three to four orders of magnitude across the numerous bird and mammal species that have been studied. Clearly, such high variability (which is not uncommon) creates difficulties in identifying representative TRVs for various taxonomic groupings of mammals and birds (i.e., individual species, genus, family, or order). In many cases, such values cannot be developed for some or all of these taxonomic groups from the available toxicology data with a high degree of confidence. Conservative TRVs that apply to broad taxonomic classes allows for greater use of the available animal toxicology data for a given substance and reduces some of the uncertainty inherent in developing TRVs for more specific taxonomic groupings. These types of TRVs are often considered appropriate for initial quantitative ERAs risk estimates (which is the case herein).

Because the TRVs presented in **Table 5-14** are initial conservative values, calculated exposures that exceed these TRVs should not be considered a definitive indication of ecological risk. If ecological risks were predicted in the ERA using these initial TRVs, alternate (or secondary) TRVs could be utilized. The data compiled in U.S. EPA (2007) also allows for further refinement of TRVs for specific ROCs, if/as necessary. The US EPA documentation provides tables of all NOAELs and LOAELs that were considered, organized by species. If warranted, these data may be used to develop TRVs for more specific taxonomic groupings, assuming sufficient toxicological data exist. This step was only considered if use of the alternate (secondary) TRVs resulted in predictions of ecological risk to the bird and mammal ROCs evaluated in the ERA. **Table 5-15** provides a summary of the alternate (or secondary) TRVs to be applied, as required. The TRVs were calculated based on either the geometric mean of LOAELs for reproduction, growth, and survival, if available in the U.S. EPA EcoSSL supporting documents, or by the application of the dose considered to be a chronic LOAEL (Sample et al. 1996).

Table 5-15: COPC Secondary Toxicity Reference Values for Mammalian; Avian; and Avian SAR ROCs

| COPC | Avian TRV (mg/kg-BW/d) | Mammalian TRV (mg/kg-BW/d) |
|-----------|--|--|
| Cadmium | 2.37 (lowest bounded reproductive LOAEL for starling in US EPA, 2005a) | 2.28 (lowest LOAEL; US EPA, 2005a) |
| Lead | 9.9 (EC20 for chicken in US EPA, 2005b) | 35 (geometric means of reproductive LOAELs from US EPA, 2005b, with 3-fold uncertainty factor) |
| Strontium | Insufficient data | Insufficient data |

| COPC | Avian TRV (mg/kg-BW/d) | Mammalian TRV (mg/kg-BW/d) |
|----------|---|---|
| Vanadium | 1.9 (geometric mean of bounded reproductive and growth LOAELs from US EPA, 2005c) | 6.7 (geometric mean of bounded reproductive and growth LOAELs from US EPA, 2005c) |

5.5 Risk Characterization

In an ERA, the risk characterization step is the process by which the probability 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 (FCSAP, 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 (FCSAP, 2012a; CCME, 1996).

The risk characterization of ROCs in the ERA utilized a simple qualitative weight of evidence (WOE) approach. FCSAP (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, ranging from those that are qualitative, and/or 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 (FCSAP, 2012a).

ROCs selected in the problem formulation included terrestrial vegetation and soil invertebrate communities, freshwater aquatic life communities, amphibian communities, herbivorous small mammal populations (represented by the meadow vole and snowshoe hare), carnivorous small mammals (represented by the ermine); herbivorous birds (represented by the ruffed grouse); insectivorous birds (represented by the dark-eyed junco). Various COCs were identified for these ROCs, COCs were also identified in surface water and sediment for freshwater aquatic life (**Section 5.2.8**). Quantitative exposure and risk modelling was conducted for the meadow vole, snowshoe hare, ermine, ruffed grouse and junco. A discussion of potential risks to these ROC based on site chemical concentrations is provided below.

Ecological Hazard Quotients

The ecological hazard quotient (EHQ) is calculated for receptors that are quantitatively modelled in the ERA and consists of a simple ratio between the estimated exposure rate for a given ROC (i.e., EXP_{Total}), and the applicable TRV, as follows:

$$\text{Ecological Hazard Quotient (EHQ)} = \frac{EXP_{Total} \text{ (mg/kg BW/day)}}{TRV \text{ (mg/kg BW/day)}}$$

It is standard ERA practice to utilize 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 potentially 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., ecological hazard quotients) 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 (FCSAP, 2012) notes that hazard quotients are simple ratios, and that situations where an EHQ exceeds 1.0 only indicate 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 that 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 (FCSAP, 2012). The FCSAP ERA guidance further elaborates on key items that must be considered to put EHQs in perspective and use them as meaningful lines of evidence in ERAs.

5.6 ERA Results

Risk characterization results for each of the receptor groups evaluated in the ERA are presented in **Sections 5.6.1 to 5.6.5**. A summary of the ERA results is provided in **Section 5.6.6**, with the overall conclusions of the ERA being provided in **Section 5.7**.

5.6.1 Vegetation and Soil Invertebrates

Lead and zinc were carried forward as COPCs for vegetation and soil invertebrates based on comparison of soil data to soil contact guidelines and to background concentrations and considering the degree and frequency of exceedance. For lead, 28% of samples exceeded the applicable guideline with a max concentration 50-fold over the guideline, while for zinc, 28% of samples exceeded the applicable guideline with a max concentration 8-fold over the guideline (**Table 5-8**). For zinc, the maximum background soil concentration of 500 mg/kg exceeded the guideline of 450 mg/kg. Based on the results of the comparison to vegetation and soil invertebrate health-based guidelines, it is possible there could be effects to some of these receptors if receptors were within areas of the highest concentrations. However, the exceedance of a guideline does not necessarily indicate there will be a risk or an adverse effect. The guidelines have been derived to be conservative and apply to all vegetation types. In addition, contamination within the study boundary has been present for many years, and some vegetation and soil invertebrate species present may have adapted/acclimated to site conditions over time.

To provide another line of evidence regarding whether effects on vegetation are occurring, observations and photos of the vegetation at the site and in the surrounding area were reviewed along with in addition to the vegetation outcomes of the terrestrial habitat assessment (**Appendix D; Section 3.3.2**).

The vegetation on the site consists of forested areas (dominated by conifer trees, white pine, and black spruce) as well as some swamp and wetland areas. Areas within the study boundary but outside the AECs do resemble the surrounding areas outside the study boundary in that vegetation is not stunted or unhealthy in appearance. Some stunted/stressed vegetation was observed in the wetland areas, but this was noted to be typical of wetlands in general. Exceedances over vegetation and soil invertebrate health-based direct contact guidelines are not only noted in areas with limited vegetation, but also in areas where vegetation is more plentiful (see wetland and terrestrial habitat photos in **Appendix D**).

The AECs which are dominated by waste rock (from historical activities) have noticeably less vegetation cover than surrounding areas; however, this is likely more due to the lack of available nutrients/suitable soil substrate for vegetation to grow in these areas than an effect brought about by elevated metal concentrations. For these reasons, site vegetation can be considered impacted from physical effects from historical activities (e.g., placement of tailings areas and rock piles). In some areas within the rock piles and tailings, vegetation can be seen growing back, although the growth appears to be stunted. Again, this is likely more to do with the soil being nutrient poor rather than a result of metal contamination.

A formal survey of invertebrates in soil was not conducted at the site. However, it is generally recognized that if vegetation can occur on a given site, some degree of soil invertebrate communities will also be present. Terrestrial vegetation and soil invertebrate health are often closely linked (e.g., Gillison et al., 2003), and soil ecological function is often dependent on both healthy vegetation

and soil invertebrate communities. Thus, if one of these receptor groups is adversely affected by soil contamination, the other group typically is as well. As with vegetation on site, AECs which are dominated by waste rock pilings or rocky areas as a result of historical activities likely do not contain the soil necessary to support soil invertebrate communities; however, as vegetation at the site appears largely healthy and typical for its setting, there is no reason to suspect the occurrence of adverse effects on soil invertebrate communities.

Considering the available lines of evidence and using a WOE approach, the potential for adverse effects to vegetation and soil invertebrate communities as a results of COPCs is within the study boundary is considered to be negligible to low based on the available data, within most areas of the site. In areas where waste rock and tailings have been placed, the likelihood of potential risks is medium to high. The presence of waste rock and tailing and the lack of nutrients in the “soils” in these areas, have most likely impacted vegetation in these areas and its ability to grow back/repopulate.

5.6.2 Birds and Mammals

Table 5-16 presents EHQs for the mammalian and avian ROCs that were evaluated in the ERA based on the initial TRVs from **Table 5-14**, while **Table 5-17** provides relevant EHQs for ROC evaluated with final TRVs. **Appendix K** provides further information on the exposure and risk estimates for these ROCs.

Table 5-16: Calculated EHQs for Mammalian and Avian ROCs Based Upon Initial TRVs

| COPC | EHQ | | | | | |
|-------------------|-----------------|-----------------|-----------------------|----------------|----------------|-----------------|
| | Snowshoe Hare | Meadow Vole | Common (Masked) Shrew | Ermine | Ruffed Grouse | Dark Eyed Junco |
| Target EHQ | < 1.0 | < 1.0 | <1.0 | <1.0 | <1.0 | < 1.0 |
| Cadmium | 0.11 | 0.079 | 8.5 | 0.16 | 0.13 | 1.6 |
| Lead | 3.9 | 1.4 | 57 | 5.5 | 20 | 130 |
| Strontium | 1.5 | 1.0 | 4.8 | 0.55 | 0.58 | 2.2 |
| Vanadium | 0.028 | 0.010 | 0.16 | 0.032 | 0.52 | 2.6 |

Notes:

Bold indicates that the total calculated EHQ exceeds the target EHQ.

EHQ values rounded to 2 significant figures.

Table 5-16 shows that there were elevated EHQs for lead for all ROC. Cadmium exceeded the target EHQ of 1.0 for the masked shrew and dark-eyed junco; strontium exceeded for the snowshoe hare, meadow vole, masked shrew, and dark-eyed junco; and vanadium exceeded for only the dark-eyed junco. As previously noted, these EHQs were conservatively based on no (or low) effect levels. The secondary TRVs in **Table 5-15** were therefore used for the receptors having elevated EHQs in order to refine estimates of potential ecological risk as presented in **Table 5-17**.

As previously noted, while less conservative than the initial TRVs, the alternate (secondary) TRVs (which include geometric means of LOAELs for growth and reproductive effects) remain adequately conservative and protective of the mammalian and avian ROCs evaluated in the ERA.

Table 5-17: Calculated EHQs for Mammalian and Avian ROCs Based Upon Secondary TRVs

| COPC | EHQ | | | | | |
|-------------------|-----------------|-----------------|-----------------------|-----------------|-----------------|-----------------|
| | Snowshoe Hare | Meadow Vole | Common (Masked) Shrew | Ermine | Ruffed Grouse | Dark Eyed Junco |
| Target EHQ | < 1.0 | < 1.0 | < 1.0 | < 1.0 | < 1.0 | < 1.0 |
| Cadmium | - | - | 2.9 | - | - | 1.4 |
| Lead | 0.52 | 0.19 | 7.6 | 0.74 | 3.3 | 22 |
| Strontium | NC | NC | NC | - | - | NC |
| Vanadium | - | - | - | - | - | 0.47 |

Notes:

NC denotes EHQ not calculated. Due to insufficient data to define TRVs for the applicable ROC, EHQs could not be calculated.

EHQ values rounded to 2 significant figures.

Bold indicates that the total calculated EHQ exceeds the target EHQ.

Shading and “-“ indicate that an EHQ was not calculated as the primary EHQ did not exceed the target EHQ of 1.

When EHQs are calculated based on the secondary TRVs, there were no elevated EHQ for the snowshoe hare, meadow vole or ermine indicating the likelihood of potential adverse effects in herbivorous small mammals and carnivorous mammals would be low. The ruffed grouse had an elevated EHQ for lead at 3.3, indicating a potential risk to individual grouse when in the study area if exposed. The grouse feeds on leaves, berries, and soil invertebrates. The areas having the greatest lead concentrations would have limited applicable food resources to support populations of grouse and other herbivorous birds. The masked shrew exceeded the EHQ for lead and cadmium; however, no effects-based TRV could be derived for strontium. The dark-eyed junco (an insectivorous bird species) exceeded the critical EHQ for (cadmium and lead; not enough information was available to derive a strontium LOAEL-based TRV). The dark-eyed junco did not exceed the critical EHQ for vanadium, indicating the risk of vanadium to insectivorous birds from vanadium is low. These results indicate a potential for individual level effects in these receptors.

Overall, the EHQ values presented in **Table 5-16** and **Table 5-17** suggest there could be individual level effect to insectivorous small mammals (represented by the masked shrew), herbivorous birds (represented by the ruffed grouse), and insectivorous birds (represented by the dark eyed junco). The study boundary is large (51 ha). When metals other than lead are considered, the exceedances over avian and mammalian health-based guidelines incorporates a much smaller area within the study boundary, with poor habitat in much of these areas. When lead is considered, the affected areas are more widespread. Given the size of the site, it was considered large enough (based on N_s calculations) to

potentially support small local populations of insectivorous small mammal populations, but not populations of insectivorous or herbivorous birds, or herbivorous or carnivorous small mammals.

While LOAEL-based TRVs were not available for strontium, lead was recommended for risk management measures in the HHRA (See **Section 4.8**). Elevated lead and strontium were often in the same samples, and as such, a reduction in lead in soils would likely also reduce strontium in soils. Given this, exposures to ecological receptors of lead and strontium are expected to be reduced once the risk management measured for human health are in place. It is also worth pointing out that much of the area with the highest concentrations would not be ideal habitat for some receptors and / or their food sources.

In addition, based on the limited amount of vegetation analytical data, which was collected in contaminated areas of the site in Wetland #3 (an area that was recommended for risk management in the HHRA, where lead concentrations in vegetation would be expected to be higher than elsewhere on the site), the estimated vegetation exposures have likely been substantially overestimated. Measured lead concentrations in stem and leaf tissue samples from areas of Wetland #3 that were measured or inferred to have lead soil concentrations greater than 2100 mg/kg, ranged from 0.8 mg/kg (sample 24-WL3-VEG1) to 37 mg/kg (sample 24-WL3-VEG5), with a mean concentration of 7.6 mg/kg. Five of the six analytical results for lead in stems and leaves collected from within Wetland #3 were below the estimated vegetation tissue concentration used in the food chain modeling for the ERA (i.e., 4.4 mg/kg); therefore, outcomes of the ERA modelling are considered to be reasonably accurate and protective of the ecological receptors that were assessed, in consideration of the Wetland #3 vegetation data (see **Appendix L** for further discussion).

As previously noted in **Section 5.5.1**, EHQs can provide a conservative indication of whether or not the assessed exposure conditions pose a potential for ecological risks; however, they are not necessarily definitive or even accurate representations of true ecological risk. In addition, they cannot predict the specific probability that an adverse effect will occur in a given ROC or 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 and reliable enough to merit further study or corrective action. Outcomes for the other LOE considered in the ERA of the mammalian and avian ROCs are summarized in the following bullets.

- The study boundary has a total area of roughly 51 ha which exceeds the ASTM spatial criterion of 0.81 ha. The area encompassing the study boundary is considered large enough to potentially support local populations of small mammalian receptors with small home ranges based on the ASTM spatial criterion and Ns calculations. However, only select areas within the study boundary contain elevated soil concentrations (**Figures 11 and 12, Appendix A**). The study boundary would not be anticipated to support to avian populations based on available habitat and food resources and population densities for the avian ROC;

- Review of habitat preferences for the mammalian and avian ROCs shows that the study boundary conditions are somewhat consistent with the preferred habitat types for some of these ROCs (and other small mammals and birds); albeit more limited in the central areas of the study boundary. Thus, it can be concluded that the study boundary offers suitable habitat conditions and food resources for the ROCs and other common small mammal and avian receptors. As such, it is considered likely that individual birds or mammals or limited populations of small mammals (represented by the ROCs) may occur on or utilize the land within the study boundary for foraging, nesting and/or breeding;
- Estimated N_s values (**Table 5-3**), suggest that the land within the study boundary could support local populations of meadow vole and potentially the masked shrew, and possibly the snowshoe hare. While the N_s calculation is not without uncertainty and does not account for literature reports of actual measured population density and territoriality considerations, it provides a reasonable and often conservative indication of the numbers of individual organisms that may be expected to use a habitat subdivision on a given site. As such, it would be unrealistic to assume that ROC presence and foraging occurs extensively within the study boundary, but not on the various adjacent areas that have equally suitable (or better) habitat conditions and food availability. There is nothing to distinguish the site from other areas as offering unique or preferred habitat or food resources for any known ecological receptor population or community. This further supports that use of lands within the study boundary by avian receptors would likely be limited to individuals, while it could support population of some small mammals with small home ranges;
- Based on available vegetation tissue data for stems and leaves in the contaminated areas of Wetland #3, estimated vegetation exposures are likely substantially overestimated when the entire site is considered; and
- While the lands within the study boundary appears likely to provide habitat conditions that can support populations of small mammalian ROCs, it is probably unrealistic to assume that ROC presence and foraging occurs extensively within the study boundary, but not on the various adjacent properties that have equally suitable habitat conditions and food availability. There is nothing to distinguish the lands within the study boundary from these other areas as offering unique or preferred habitat for any ecological receptor population or community. ASTM (2011) states that if a subject property'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. Given the large size of the site and given the habitat and food resources are limited within some of the central portion of the site having the highest contamination relative to other areas of the site and adjacent off-site areas, then an animal is likely to spend less of its time within the highest concentration areas for metals (see **Figure 8, Appendix A**).

Collectively, consideration of all available LOE and using a WOE approach, suggests that small mammal and avian ROCs may occur within or near the study boundary, and some individuals may be expected to spend sufficient time at this location (foraging, nesting, breeding, or resting) to incur significant exposures to COPCs in soils within the study boundary.

For masked shrew, ruffed grouse, and dark-eyed junco, critical EHQ values were exceeded when using effects-based TRVs, and as such there could be potential effects in individuals if they are in the areas and obtain food resources where elevated concentrations occur. However, based on the size of the study boundary and available habitat and food resources, and consideration of population densities of avian ROC, there is a low likelihood for population level effects in the grouse or junco or other herbivorous or insectivorous birds. The likelihood of potential effects on local populations of small herbivorous small mammals, herbivorous birds or insectivorous birds is considered to be negligible to low. Given the home and foraging range for the shrew would be encompassed by the areas having elevated metals, and based on the elevated effects-based EHQs, there is a low to medium potential for effects on local populations of insectivorous small mammals with small home ranges.

Some miscellaneous drums and debris were noted by AEC 1 and south of AEC 12. Some of the drums were partially buried and seemed empty. While not a chemical exposure issue, these drums and debris could pose a physical risk to wildlife in the area. It would be prudent to remove the visible debris and drums.

The approaches and assumptions used to calculate EHQs for the mammalian and avian ROCs is considered to be conservative and likely substantially overestimate the potential for ecological risk. The major areas of conservatism in the ERA are described and discussed in **Section 5.7**.

The planned risk management/remediation for the study boundary based on the HHRA outcomes (**Section 4.6**), will also reduce lead concentrations in soil that ecological receptors would be exposures to, and hence would also reduce potential risks.

5.6.3 Freshwater Aquatic Life

The assessment of potential risks to freshwater aquatic life ROCs followed a simple qualitative weight of evidence (WOE) approach using the available lines of evidence which included the outcomes of the comparison of surface water and sediment to freshwater aquatic life guidelines and outcomes of the aquatic habitat assessment, the benthic community investigation, and the aquatic vegetation and bivalve investigations. Further details pertaining to the benthic, vegetation, and bivalve investigations can be found in **Appendix L**.

5.6.3.1

Surface Water

No COPCs were identified in Lake Enon surface waters based on the outcomes of the comparison of site surface water data to freshwater aquatic life health-based groundwater quality benchmarks. While the settling/tailings ponds were not originally fish habitat and were constructed for site activities, fish have been observed in some of these areas. There were numerous minnows observed in the tailings pond (AEC 6, which has an outlet that flows into Lake Enon), while the settling pond (AEC 7, which is also connected to Lake Enon) contained small fish and minnows.

An additional line of evidence for freshwater aquatic life was the outcome of the aquatic habitat assessment. Four watercourses were identified within the study boundary and assessed as part of the aquatic habitat assessment. Two of the watercourses were confirmed to have fish, one was considered to provide seasonally accessible fish habitat and the other was considered unlikely to provide fish habitat. Additional details are provided below.

WC1 is a permanent watercourse and drainage feature of the Site. Although fish were observed in this stream which flows northward into a culvert towards a human-made settling pond (AEC 7), fish passage is prevented upstream of this watercourse by rock and boulder infill (**Appendix D**). The stream is considered to provide suitable habitat for fish based on direct observations of fish; however, this watercourse is considered to not provide optimal browsing or rearing habitat for fish due to limited stream cover and passage to upstream watercourses. In May 2024, WC1 was reported to have a mid-channel depth of 0.04 m and wet width of 1.0 m (Envirosphere, 2024; **Appendix L**). In July 2023, WC1 was reported to have a depth of 0.3 m and a wet width of 3.0 m (**Appendix D**).

WC 2 initiates from the outlet for a settling pond (AEC 7) and flows into Lake It Enon. This watercourse is expected to support fish based on observations of fish in the settling pond. The boulder substrate is unlikely to support breeding; however, the stream is likely to support fish passage and foraging between Lake Enon and the human-made settling ponds (**Appendix D**). In May 2024, WC2 was reported to have a mid-channel depth of 0.03 m and a wet width of 0.3 m (Envirosphere, 2024; **Appendix L**). In July 2023, WC2 was reported to have a depth of 0.25 m and a wet width of 2.0 m (**Appendix D**).

WC3 initiates at the outlet for a settling pond (AEC 6) and flows into Lake Enon. The stream is likely to support fish passage between Lake Enon and the human-made settling pond. The narrow stream is surrounded by tall grass which could provide suitable cover and foraging for small or juvenile fish during some times of the year (**Appendix D**). In May 2024, WC3 was reported to have a mid channel depth of 0.18 m and a wet width of 0.31 m (Envirosphere, 2024; **Appendix L**). In July 2023, WC3 was reported to have a depth of 0.2 m and a wet width of 0.5 m (**Appendix D**).

Cadmium, lead, and zinc were identified as COPCs for freshwater aquatic life in these on-site watercourses, based on a limited number of surface water samples collected at one point in time. While these watercourses may offer habitat to freshwater aquatic life, they are unlikely to provide breeding

habitat for these receptors. Given the small size of these water courses, the varying degree of water presence (as noted by the recorded water depths and wet widths), and given the water course information presented above, it is unlikely that these water course can support permanent pelagic freshwater aquatic life communities. As such, even though some metals had measured surface water concentrations in excess of freshwater aquatic life guidelines, the likelihood that these concentrations would result in community level effects in freshwater aquatic life is considered to be low.

5.6.3.2 Groundwater

Several chemicals were selected for further evaluation for freshwater aquatic life based on concentrations in groundwater exceeding their respective freshwater aquatic life-based guideline (i.e., the FIGQGs (FCSAP, 2016); MECP, 2011, groundwater, shallow, potable). These included cadmium, copper, lead, nickel, selenium, and zinc. No guideline was available for strontium in groundwater, but it is considered to be site related.

Based on the screening of the groundwater concentrations to applicable benchmarks, exceedances were noted for cadmium (11 of 11 samples), copper (2 of 11 samples), lead (4 of 11 samples), nickel (2 of 11 samples), selenium (3 of 11 samples), and zinc (5 of 11 samples).

The applied ground water guidelines for cadmium, copper, lead, nickel, and selenium (FIGQGs) are based on generic modifying factors (e.g., hardness, pH, DOC). No site-specific water quality parameters were available to adjust the FIGQG to specific site conditions. However, none of these metals (including strontium) exceeded the surface water quality guidelines for freshwater aquatic life in Lake Enon, and concentrations were either similar to background or not detected.

Metals are typically found in groundwater and concentrations decline significantly when oxygen-deprived groundwater discharges to the oxygen-rich surface water environment resulting in the precipitation of iron oxyhydroxides (and co-precipitation of various other metals). As such, the likelihood of groundwater concentrations of the COPCs to represent a concern to freshwater aquatic life is considered to be low.

5.6.3.3 Sediment

Guideline Exceedances

Several chemicals were selected as COCs for freshwater aquatic life based on concentrations in sediment exceeding their respective freshwater aquatic life-based guideline, including the following:

- Cadmium in 31 of 43 samples, exceeding the guideline of 3.5 mg/kg by a maximum of 6.6-fold;
- Lead in 34 of 43 samples, exceeding the guideline of 91.3 mg/kg by a maximum of 62-fold;
- Selenium in 7 of 43 samples, exceeding the guideline of 2.0 mg/kg by a maximum of 4.4-fold,
- Silver in 36 of 43 samples, exceeding the guideline of 0.5 mg/kg by a maximum of 5.6-fold,
- Zinc in 36 of 43 samples, exceeding the guideline of 315 mg/kg by a maximum of 9.8-fold;

- Modified TPH (all resembling lube oil) in 4 of 7 samples, exceeding their respective TOC-adjusted guidelines by a maximum of 6.4-fold; and
- Strontium (no guideline available but carried forward as strontium concentrations are likely related to historical activities at the site).

There was no clear gradient of the degree of sediment guideline exceedance as samples moved further from the study boundary, although the lead exceedances were typically elevated closer to the study boundary that borders Lake Enon. Although there were several guideline exceedances for metals observed in site sediments and sediments in Lake Enon, the exceedance of a guideline does not necessarily indicate that an adverse effect will occur as the guidelines are conservative in nature. Published regulatory benchmarks are generic values, with no consideration given to site-specific populations/communities or media conditions that influence bioavailability. Exceedance of these values does not necessarily imply there is a risk of adverse effects; rather, it suggests that further evaluation is warranted (such as further ecological risk assessment or data collection). As previously noted, this is especially true for metals, many of which have essential nutritional and physiological roles in terrestrial biota. The acclimation and adaptation to metals is well known to occur (e.g., Kapustka et al., (2004), ICM (2007), Chapman (2008), and Newman and Clements (2008)) and should be considered. Given the contamination has been present for decades, it would be likely that some acclimation or adaptation of species has occurred.

In addition, several of the sediment samples within Lake Enon had total organic carbon ranging from 1% to 5% (**Appendix E**). The high organic carbon could reduce the bioavailability of metals for benthic species within Lake Enon.

Some exceedances of modified TPH exceedances over their respective TOC-adjusted guideline occurred in the settling/tailings pond samples (SED4, 5, 8, and 9) within AEC 6 and AEC 7. TOC concentrations ranged from 4,500 mg/kg at SED12 to 56,000 mg/kg at SED17. However, the settling/tailings ponds were not originally habitat for aquatic life; rather, they are a result of aquatic life being able to access these areas after they were created.

Benthic Invertebrate Community

The benthic invertebrate community data provide key insights into the potential ecological risks of site-related contamination. As discussed in **Appendix L**, the cluster analysis indicated significant differences in taxonomic group abundance between watercourse and Lake Enon sample locations, which is expected as these areas represent two distinct ecological environments.

Despite metal exceedances present in sediment samples at all three watercourse sites, Hilsenhoff Biotic Index (HBI) results indicate that water quality remains fair at WC-2 and good at WC-1 and WC-3, suggesting limited impacts on oxygen levels from these exceedances. It should be noted that the HBI was designed as a measure of organic stream pollution and tolerance values for each taxon were based

on their tolerance to low oxygen environments (Hilsenhoff, 1987), rather than metals concentrations in sediments. However, HBI scores can still reflect overall tolerance to stressors for benthic taxa.

The differences observed in benthic communities might arise from other biophysical factors at each sampling site rather than metals contamination, such as: proximity or connectivity to Lake Enon predation, habitat characteristics and substrate types, ecological dynamics, and the general stochastic nature of ecological communities.

Within Lake Enon, benthic communities showed varied similarities. The HBI scores generally indicated poor water quality throughout the lake, with the exception of INV-07 (which is located to the south of the lake and close to the former mine site), which recorded fair quality. EPT ratios were notably low within the lake sites, which is expected as EPT presence is generally more common in flowing waters (e.g., streams) than in lentic environments.

While there are sediment metal exceedances in the lake, a correlation between these exceedances and benthic community structure was not observed. For example, sample INV-06, which demonstrated the lowest contaminant concentrations, also demonstrated low abundance and taxa richness.

In conclusion, the evaluation of indices such as abundance, taxon richness, EPT ratio, HBI, and the Bray-Curtis Index revealed poor correlation between contamination levels and benthic community characteristics in Lake Enon or the watercourses. These data suggest that other factors than sediment contamination, such as substrate type, predation, connectivity, and general ecological variability may play significant roles in shaping community assemblages.

Aquatic Vegetation and Bivalves

The analysis of aquatic vegetation and bivalve tissue data provides an additional line of evidence in evaluating the potential ecological risks posed by site-related contaminants in the sediments of Lake Enon. As detailed in **Appendix L**, a total of three far-field samples and nine near-field samples were collected for both aquatic vegetation and bivalve tissues, with paired sediment samples also obtained to determine co-located sediment concentrations of COPCs.

The analytical results in **Appendix L** revealed the presence of various COPCs in both near-field and far-field samples. For aquatic vegetation, cadmium, lead, strontium, and zinc were detected in all near-field and far-field samples, with near-field concentrations occasionally exceeding those from the far-field locations but only marginally. Bivalves tissue concentrations showed a similar pattern, but all COPCs identified in sediment were detected in this media type (i.e., cadmium, lead, selenium, silver, strontium, and zinc). Similar to vegetation, near-field bivalve tissue concentrations were generally comparable to or only marginally above far-field concentrations. To assess the relationship between sediment concentrations and the levels of COPCs in aquatic vegetation and bivalves, the data were evaluated for linear correlation. The resulting R-squared values for each COPC were found to be low (generally less

than 0.1), indicating a weak linear correlation between sediment concentrations and those found in biological tissues. This suggests that the concentrations of COPCs in the aquatic vegetation and bivalve tissues are unlikely to be solely attributable to site-related sediment contamination. Furthermore, locations where elevated tissue concentrations were observed did not consistently align with higher sediment concentrations (i.e., poor correlation), indicating that factors other than sediment-derived contaminants are likely influencing tissue accumulation.

Given these findings, it appears that while there are detectable levels of various COPCs in aquatic vegetation and bivalve tissues from Lake Enon, the risk to these organisms from site-related sediment contamination appears to be low. This conclusion is supported by the observations that aquatic vegetation and bivalves were relatively abundant in the areas where substrate conditions are suitable, and do not appear to be showing signs of impairment.

5.6.3.4

WOE Conclusion for Freshwater Aquatic Life

Although cadmium, lead, and zinc were identified as COCs for freshwater aquatic life in on-site watercourses that may offer habitat for these species groups, it is unlikely that these environments provide breeding habitat for these receptors. Given the small size of these water courses and the varying degree of water present, it is unlikely that these watercourses can support pelagic freshwater aquatic life communities. As such, even though some metals had measured surface water concentrations in excess of freshwater aquatic life guidelines, the likelihood that these concentrations would result in community level effects in freshwater aquatic life is considered to be low.

With respect to groundwater, metals are typically found in groundwater and concentrations decline significantly when oxygen-deprived groundwater discharges to the oxygen-rich surface water environment resulting in the precipitation of iron oxyhydroxides (and co-precipitation of various other metals). As such, the likelihood of groundwater concentrations of the COPCs to represent a concern to freshwater aquatic life is considered to be low.

For benthic species, several chemicals were selected as COCs for freshwater aquatic life based on concentrations in sediment exceeding their respective freshwater aquatic life-based guideline. The evaluation of benthic community indices such as abundance, taxon richness, EPT ratio, HBI, and the Bray-Curtis Index revealed no strong correlation between contamination levels and benthic community characteristics in Lake Enon or the watercourses. These data suggest that other environmental parameters, such as substrate type, predation, connectivity, and general ecological variability may play more significant roles in shaping community assemblages than site-related contamination. In addition, the relatively similar sediment COC concentrations in near-field and far-field aquatic vegetation and bivalve tissues, along with the poor observed relationship between sediment samples co-located at vegetation and bivalve sample locations, provides an additional line of evidence to support that the risk to these organisms from site-related sediment contamination is low.

Considering the available lines of evidence and using a WOE approach, the potential for adverse effects in pelagic-feeding freshwater aquatic life communities as a result of contamination on the site is considered to be negligible based on the available data, and the potential for adverse effects to benthic species is considered to be low.

5.6.4 Amphibians

The assessment of potential risks to amphibians followed a simple qualitative weight of evidence (WOE) approach using the available lines of evidence, which included: the outcomes of the comparison of surface water and sediment to freshwater aquatic life guidelines and amphibian toxicity data; comparison of the spatial size and habitat quality to surrounding areas; receptor habitat preferences; and the outcomes of the wetland habitat assessment.

5.6.4.1 Surface Water

Based on the screening of the wetland surface water concentrations to freshwater aquatic life benchmarks, exceedances were noted for aluminium (1 of 9 samples; Wetland #1-B only), cadmium (1 of 9 samples; Wetland #1-B only), iron (7 of 9 samples; Wetland #1-B; Wetland #2; Wetland #3; 24-WL3-SW1; 24-WL3-SW3; 24-WL3-SW4; 24-WL3-DUPA), lead (7 of 9 samples; Wetland #1-B; Wetland #2; Wetland #3; 24-WL3-SW2; 24-WL3-SW3; 24-WL3-SW4; 24-WL3-DUPA), manganese (6 of 9 samples; Wetland #2; Wetland #3; 24-WL3-SW1; 24-WL3-SW3; 24-WL3-SW4; 24-WL3-DUPA), selenium (1 of 4 samples; Wetland #1-B only), 7 of 9 samples for zinc (see **Table J-2, Appendix J**). While iron and manganese exceeded guidelines in surface water, as discussed previously these parameters are likely related more to background conditions than site contamination. As such, iron and manganese were not carried forward for further assessment, while cadmium, lead, selenium, and zinc were carried forward. No COPCs were identified for the surface water of Lake Enon based on the available data.

Wetland sample #1-B had a measured cadmium concentration of 0.00012 mg/L which exceeds the freshwater aquatic life guideline of 0.00009 mg/L by 1.3-fold. This guideline is based on an assumed water hardness of 50 mg/L CaCO₃ as hardness data for the wetland sample in question was not available. Surface water toxicity data for amphibians has been compiled in FCSAP (2019) for several metals, including cadmium, lead, and zinc. For cadmium, one study reported an effect level of approximately 20% at a test concentration an order of magnitude greater than the maximum surface water concentration observed in the wetland samples and our site maximum concentration, and the majority of studies reported effects on less than 10% of the population at a concentration of this magnitude (FCSAP, 2019).

Lead concentrations in wetland surface waters were reported at levels of <0.0005 mg/L, 0.00056 mg/L, 0.0018 mg/L, 0.0017 mg/L, 0.0027 mg/L, 0.0036 mg/L, 0.0048 mg/L, 0.008 mg/L and 0.033 mg/L. These data indicate that lead concentrations at our site exceed the established guideline for the protection of freshwater aquatic life of 0.001 mg/L by 2.7-fold to 33-fold. Site-specific pH data were not available, and therefore the guideline has not been adjusted based on pH. The available amphibian surface water

toxicity data (FCSAP, 2019) demonstrate that observed adverse effects on amphibians are typically associated with considerably higher lead concentrations than the maximum concentration observed within the wetlands sampled. Generally, no significant adverse effects on amphibians were observed at lead concentrations as high as 0.01 mg/L, with effect levels remaining below 20% in most studies up to a concentration of 0.1 mg/L.

Zinc concentrations in wetland surface waters ranged from 0.0078 mg/L to 0.052 mg/L, exceeding the guideline for the protection of freshwater aquatic life of 0.001 mg/L by up to 74-fold. However, adverse effects on amphibians exposed to zinc are typically associated with considerably higher zinc concentrations than the maximum concentration observed at the site. The majority of studies compiled in FCSAP (2019) did not report significant adverse effect levels on amphibians at zinc concentrations in the same order of magnitude the maximum site wetland concentration; in fact, effect levels of greater than 20% were reported in the majority of studies at concentrations of at least 1 mg/L.

Among the nine wetland samples, Wetland #1-B reported a slightly elevated selenium concentration of 0.0011 mg/L, exceeding the selenium guideline of 0.001 mg/kg by a factor of 1.1. However, this maximum site concentration is significantly lower than the lowest observed effect concentration (LOEC) for amphibians reported in Lee (1978), which is documented at 0.25 mg/L and based on endpoints related to morphology and length. Additionally, the CCME guideline is essentially equal to the maximum reported site concentration. Therefore, it is not anticipated that maximum observed site concentrations of selenium would result in adverse effects to populations of amphibians accessing the site.

The potential impact on amphibians in surface waters within the study boundary wetlands and off-site in Lake Enon appears to be relatively low compared to the concentrations at which adverse effects were observed in amphibians in the broader body of toxicity studies documented in FCSAP (2019) and other sources.

Outcomes of the wetland habitat assessment indicated the presence of suitable habitat for amphibians within the study boundary (**Appendix D**). There are also other lakes outside of the study boundary that would offer habitat that is of similar or better quality as habitat within the study boundary.

In summary, the likelihood the concentrations of COPCs in measured wetland and Lake Enon surface water to pose a risk to area amphibians is considered to be low based on the available data.

5.6.4.2

Sediment

Sediment data in the wetlands were compared to the Canadian sediment guidelines for the protection of freshwater life to identify COPCs (i.e., cadmium, lead, silver, and zinc). The sediment data for the identified COPCs were also compared to available amphibian sediment toxicity data compiled by FCSAP (2019) and other available sources.

The maximum cadmium concentrations in wetland sediments were reported at 3.8 mg/kg, slightly exceeding the guideline for the protection of freshwater aquatic life of 3.5 mg/kg. While there is a lack of background sediment data representative of wetland habitats in the current study, it is worth noting that sediment data from the background location in Lake Enon reported a cadmium concentration of 1.2 mg/kg. Amphibian sediment toxicity data compiled in FCSAP (2019) report adverse effects on 20 – 40% of amphibians after exposure to cadmium concentrations in sediment in the range of the maximum measured wetland concentration as well as the background Lake Enon sediment concentration. These data suggest that concentrations of cadmium in wetlands within the study boundary fall within the range of potential effects; however, the background sample also fell within this range. This suggests that amphibians may not face a significantly higher risk of cadmium exposure due to site activities when compared to the potential risks associated with natural background concentrations.

Lead exceeded the freshwater aquatic life sediment guideline of 91.3 mg/kg in all wetland samples. Sediment concentrations collected from samples in the 3 of the 10 wetland samples were only slightly above the sediment guideline at 100 mg/kg, 280 mg/kg and 120 mg/kg; however, in Wetland #3, lead in sediment was reported at 6,100 mg/kg, 7,100 mg/kg, 9,700 mg/kg, 10,000 mg/kg, 13,000 mg/kg and 14,000 mg/kg. While sediment toxicity data for amphibians were limited, some data were identified for lead. As such, lead sediment concentrations were compared to amphibian lead toxicity data identified in the literature reviewed. At lead concentrations of 540 mg/kg, inhibitory effects (i.e., reduced body mass at metamorphosis) were reported in the southern leopard frog (*Lithobates sphenoccephalus* or *Rana sphenoccephala*) (Sparling et al., 2006). An LC25 and LC50 for the northern leopard frog (*Lithobates pipiens* or *Rana pipiens*) were reported to be 3,550 mg/kg and 4,662 mg/kg (Bleiler et al., 2004). Lead concentrations at Wetland #3 are greater than the reported effect levels identified in these studies and have the potential to pose a risk to amphibians if they are using this area of the site. During the site visit, two species of frog were observed near the wetland with elevated lead concentrations (pickerel frog in AEC 6 and wood frog in AEC 10; see habitat assessment report in **Appendix D**).

Sediment toxicity data for amphibians for silver were not identified; however, this chemical is not considered to be site related.

While direct toxicity data for strontium in sediments on amphibians is scarce, studies have shown that sediment-dwelling organisms can be affected by strontium. For instance, Borgmann et al. (2005) reported that the amphipod *Hyaella azteca*, often used as an indicator species for sediment toxicity, showed reduced growth at sediment strontium concentrations of 1,500 mg/kg. Strontium has also been shown to interfere with calcium metabolism and bone development in vertebrates (Watts and Howe, 2010), which could be especially detrimental to metamorphosing amphibians. Given the maximum strontium concentration of 5,400 mg/kg in Lake Enon wetland sediments, strontium may pose a risk to amphibians. This concentration substantially exceeds the levels associated with adverse effects in other aquatic organisms, suggesting potential risk to amphibians.

Adverse effects on amphibians exposed to zinc in sediments are typically associated with considerably higher zinc concentrations than the maximum concentration observed in sediments within the study boundary. The majority of studies compiled in FCSAP (2019) did not report significant adverse effect levels on amphibians at zinc concentrations in the same order of magnitude as the site maximum; in fact, effect levels of greater than 20% were reported in the majority of studies at concentrations of at least 1,000 mg/kg.

In summary, while the likelihood that concentrations of site-related cadmium, silver, and zinc in site wetland sediments pose potential risks to amphibians is considered low, the measured lead concentration in the sediments of Wetland #3 (the largest identified wetland) were greater than effect levels for lead in the one sample collected. Given this, and considering the available habitat and surrounding areas, the likelihood for potential effects in amphibians as a result of wetland sediments is considered to be low to medium based on the available data.

5.6.4.3 WOE Conclusion for Amphibians

Outcomes of the wetland habitat assessment indicated the presence of suitable habitat for amphibians within the study boundary (**Appendix D**); however, amphibians were not observed in Wetland #3 during field work completed at the site. A small area of standing water and cattails that was observed could potentially provide a suitable habitat for frogs; however, the majority of Wetland #3 lacks ideal habitat conditions for amphibians in general. Additionally, since the wetland is likely to freeze over during winter, it is unlikely that frogs would overwinter in this area. There are also other lakes outside of the study boundary that would offer habitat that is of similar or better quality as habitat within the study boundary.

The potential impact on amphibians in surface waters within the study boundary wetlands and off-site in Lake Enon appears to be relatively low compared to the concentrations at which adverse effects were observed in amphibians in the broader body of toxicity studies documented in FCSAP (2019) and other sources. In addition, while the likelihood that concentrations of site-related cadmium, silver, and zinc in site wetland sediments pose potential risks to amphibians is considered low, the measured lead concentration in the sediments of Wetland #3 (the largest identified wetland) were greater than effect levels for lead in the one sample collected. Given this, and considering the available habitat and surrounding areas, the likelihood for potential effects in amphibians as a result of wetland sediments is considered to be low to medium based on the available data.

5.6.5 SAR (Snapping Turtle)

The snapping turtle was identified as a SAR that could potentially be on site (**Section 5.2.4**) and potentially suitable wintering and nesting habitat was identified during the habitat assessment around AECs 6 and some areas of AEC 8 (Lake Enon). The snapping turtle is an omnivorous and opportunistic feeder, whose food items can include vascular plants, mollusks, arthropods, fish, and amphibians; however, they generally consume more vegetation than animal matter (Ernst and Lovich, 2009; ECCC,

2020a). Based on their diet and the potential for nesting and overwintering habitat in AECs 6 and 8, the snapping turtle could potentially be exposed to site-related contamination.

The snapping turtle is a reptile. Even though reptiles appear to be declining globally, significant data deficiencies have been acknowledged (Weir et al., 2014) and there remains a paucity of toxicity data and standard testing methodologies for including reptiles in ERAs.

Toxicity data from avian studies have often been used as surrogate data for land-based exposures of reptiles in ERAs. However, recent research has suggested that reptiles may have significantly different exposure profiles and their sensitivity to contaminants may be significantly different than avian species, resulting in an underestimation of potential risks (Ortiz-Santaliestra et al., 2018; Weir et al., 2014; Weir et al., 2010). A recent review of pesticide toxicity data found that warm-blooded vertebrates (i.e., birds and mammals) are generally not good surrogates for reptile and terrestrial amphibian risk assessment (Ortiz-Santaliestra et al., 2018).

Ecological risk assessments typically focus on dietary exposures. However, dermal exposures may be more important to reptiles than oral exposures (depending on the chemical), as the cold-blooded physiology of reptiles results in lower energetic demands and lower dietary exposure compared to birds and mammals (Weir et al., 2014). In addition, a high percentage of reptile body surface area is potentially in contact with contaminated media relative to other receptors.

Given the many uncertainties still remaining, the evaluation of the snapping turtle in the current ERA focussed on the potential for this species to be exposed to COPCs on the site using a WOE approach.

Suitable habitat for the snapping turtle includes slow-moving water with a soft mud bottom and dense aquatic vegetation (ECCC, 2020a). Overwintering sites require water that is shallow enough for the turtle to reach the surface to breathe, but deep enough that it will not freeze to the bottom. In addition, mud deep enough for the turtle to bury itself is a key requirement (Meeks and Ultsch, 1990; ECCC, 2020a). Eggs are generally laid on sand or gravel banks near the water in locations where vegetation is absent or sparse (ECCC, 2020a).

The site likely provides a movement corridor for the snapping turtle and may provide some habitat that fits the description for overwintering and nesting, although it appears such areas would be limited in size. Wetland areas within the study boundary were identified in the AEC 6 in late July 2023 (which followed a period of heavy rains earlier in the month). These wetlands were identified as WL-1, WL-2 and WL-3 (**Figure 4, Appendix D**) and they had only 2.5 cm, 5 cm and 2.5 cm of standing water, respectively, at the time of the assessment; therefore, most of these areas would be expected to freeze fully in the winter which may limit their use as habitat for overwintering. In addition, the areas of Lake Enon that provide potentially suitable nesting and overwintering habitat are relatively small in size (see Habitat Assessment Report in **Appendix D** and site photos in **Appendix B**), and there is no reason to

assume that snapping turtles would preferentially select the contaminated areas of Lake Enon for these activities over similar and potentially more suitable habitat in other adjacent areas of the lake. In addition, given that there are other lakes and waterbodies outside of the study boundary, it is most likely that there would be other suitable (and possibly better) habitat for the snapping turtle outside of the study boundary.

The photos of the sediment sampling locations in Lake Enon from the habitat assessment (**Appendix D**) show that vegetation close to the shore is sparse to limited, at best. In addition, grain size analysis undertaken at twelve sediment sampling locations within the settling/tailings ponds and in Lake Enon indicated most locations consisted of fine-grained sediment (i.e., 8 out of 12) and that coarse-grained sediment (i.e., SED12, SED14, SED23 and SED25) was located near the shoreline where the turtle would likely overwinter. There may be some areas within the study boundary that could be suitable for laying eggs; however, the available suitable habitat for this activity within the study boundary is not better or larger than other suitable (or better) habitat adjacent to the site.

Lastly, given that snapping turtles are opportunistic omnivores as discussed in **Section 3.4**, they have a varied diet which reduces their reliance on any single food source that may potentially be contaminated. The results of the aquatic vegetation investigation also demonstrate that there are only marginal differences in contaminant concentrations between near-field and far-field locations, suggesting that vegetation that would be consumed by this species is unlikely to be significantly impacted by site-related contaminants.

While the snapping turtle could potentially be exposed to site-related contamination, their exposures would likely be limited due to the limited suitable habitat and food resources. It is also unknown if the snapping turtle even currently occurs within the study boundary as none were observed during field work. As such, potential exposures of site-related contaminants to the snapping turtle are anticipated to be limited.

5.6.6

Summary of ERA Results

Considering the available lines of evidence and using a WOE approach, the potential for adverse effects to vegetation and soil invertebrates communities as a result of COPCs is within the study boundary is considered to be negligible to low based on the available data, within most areas of the site. In areas where waste rock and tailings have been placed, the likelihood of potential risks is medium to high. The presence of waste rock and tailing and the lack of nutrients in the “soils” in these areas, have most likely impacted vegetation in these areas and its ability to grow back/repopulate.

Collectively, consideration of all available LOE and using a WOE approach, suggests that small mammal and avian ROCs may occur within or near the study boundary, and some individuals may be expected to spend sufficient time at this location (foraging, nesting, breeding, or resting) to incur significant exposures to COPCs in soils within the study boundary. For the masked shrew, ruffed grouse, and dark-

eyed junco, critical EHQ values were exceeded when using effects-based TRVs, and as such there could be potential effects in individuals if they are in the areas and obtain food resources where elevated concentrations occur. However, based on the size of the study boundary and available habitat and food resources, and consideration of population densities of avian ROC, there is a low likelihood for population level effects in the grouse or junco or other herbivorous or insectivorous birds. The likelihood of potential effects on local populations of small herbivorous birds, insectivorous birds is considered to be low. Given the home and foraging range for the shrew would be encompassed by the areas having elevated metals, and based on the elevated effects-based EHQs, there is a low to medium potential for effects on local populations of insectivorous small mammals with small home ranges (e.g., shrew).

Considering the available lines of evidence and using a WOE approach, the potential for adverse effects in pelagic freshwater aquatic life communities as a result of contamination on the site is considered to be negligible based on the available data.

For benthic species, several chemicals were selected as COCs for freshwater aquatic life based on concentrations in sediment exceeding their respective freshwater aquatic life-based guideline. The evaluation of benthic community indices such as abundance, taxon richness, EPT ratio, HBI, and the Bray-Curtis Index revealed no strong correlation between contamination levels and benthic community characteristics in Lake Enon or the watercourses. These data suggest that other environmental parameters, such as substrate type, predation, connectivity, and general ecological variability may play more significant roles in shaping community assemblages than site-related contamination. In addition, the relatively similar sediment COC concentrations in near-field and far-field aquatic vegetation and bivalve tissues, along with the poor observed relationship between sediment samples co-located at vegetation and bivalve sample locations, provides an additional line of evidence to support that the risk to these organisms from site-related sediment contamination is low.

Based on the available site and toxicity data related to wetland and Lake Enon surface water as well as wetland sediments, the likelihood of potential risk to amphibians is considered to be low to medium, based on the available data.

While the snapping turtle could potentially be exposed to site-related contamination, their exposures would likely be limited due to the limited suitable habitat and food resources. In addition, the aquatic vegetation investigation revealed only marginal differences between COPC concentrations in near-field vegetation relative to far-field vegetation; therefore, potential uptake of site-related COPCs from vegetation ingestion is expected to be limited. It is also unknown if the snapping turtle even currently occurs within the study boundary as none were observed during field work.

Table 5-18 provides a summary of the conclusions for the ERA for the ROC based on the LOE and overall WOE.

Table 5-18: Overall Weight of Evidence (WOE) Evaluation for Each Assessed Line of Evidence (LOE) for the Receptors Evaluated in the ERA

| Receptor of Concern (ROC) | Line of Evidence (LOE) | Degree to which the LOE suggests a Potential for Effects or Impairment | Confidence in the LOE ¹ | Potential for Adverse Health Effects ¹ |
|---|--|--|------------------------------------|--|
| Terrestrial Vegetation Communities | Outcomes of the comparison of site soil chemical concentrations to vegetation health-based soil quality benchmarks and reference areas. | Medium | Low | Negligible to Low in most areas; Medium to High in areas covered with mine wastes and tailings (due physical impacts) |
| | Comparison of vegetation on site to vegetation present in the surrounding area and results of terrestrial habitat assessment. | Negligible to Low, although Medium to High in areas covered with mine waste and tailings | High | |
| | Outcomes of visual vegetation observations and outcomes of the terrestrial habitat assessment. | Negligible to Low in areas where there is less physical disturbance of soils; Medium to High in areas covered with mine wastes and tailings | High | |
| Soil Invertebrates Communities | Outcomes of the comparison of site soil chemical concentrations to soil invertebrate health-based soil quality benchmarks and reference areas. | Negligible to Low, although Medium to High in areas covered with mine waste and tailings | Low | Negligible to Low in most areas; Medium to High in areas covered with mine wastes and tailings (due physical impacts) |
| Herbivorous, Insectivorous/Invertebrates, and Carnivorous Small Mammals Populations | Outcomes of the comparison of site soil chemical concentrations to avian and mammalian health-based soil quality benchmarks. | High | Low | Negligible to Low (for small herbivorous mammals, carnivorous small mammals, herbivorous birds or insectivorous birds) Low to Medium (insectivorous small mammals with small home ranges) |
| | Outcomes of the comparison of Wetland #3 vegetation tissue data to exposure point concentration used in exposure modelling. | Low | High | |
| | Comparisons between estimated COC exposures and TRVs (i.e., ecological hazard quotients). | Low (small herbivorous mammals with moderate to large home ranges; carnivorous small mammals; herbivorous and insectivorous birds); Low to Medium (herbivorous and insectivorous small mammals with small home ranges) | Medium | |

| Receptor of Concern (ROC) | Line of Evidence (LOE) | Degree to which the LOE suggests a Potential for Effects or Impairment | Confidence in the LOE ¹ | Potential for Adverse Health Effects ¹ |
|---|--|---|------------------------------------|--|
| | Outcomes of the consideration of terrestrial habitat assessment; spatial size of available habitat within and adjacent to the study boundary, and receptor home ranges and consideration of Ns for likelihood of population level effects. | Negligible (avian species and mammals with moderate to large home ranges); Low to Medium (small mammals with small home ranges) | Medium | |
| Freshwater Pelagic and Benthic Aquatic Life Communities | Outcomes of the comparison of site surface water and groundwater to freshwater aquatic life health-based benchmarks for surface water and groundwater. | Negligible | Low | Negligible for pelagic feeding; low for benthic species |
| | Outcomes of the comparison of site sediment concentrations to freshwater aquatic life health-based sediment benchmarks. | High | Low | |
| | Outcomes of the aquatic habitat assessment. | Low | High | |
| | Outcomes of the benthic community, aquatic vegetation, and bivalve investigation | Low | Medium | |
| Amphibian Communities | Comparison of freshwater aquatic life toxicity data and amphibian toxicity data to site concentrations. | Medium | Low | Low to Medium |
| | Comparison spatial size and habitat quality to surrounding areas, comparison of receptor habitat preferences to subject property habitat conditions and habitat quality. | Low to Medium | Medium | |
| | Outcomes of the wetland habitat assessment. | Low | Medium | |
| SAR (Snapping Turtle) | Qualitative outcomes of the exposure potential via the relevant exposure routes. | Due to limited toxicity data, potential risk to the snapping were not evaluated; however, exposure potential was considered. | | Site-related exposures to snapping turtles would likely be limited due to the limited suitable habitat and food resources, as well as relatively low COPC concentrations in food sources that have been analyzed (i.e., aquatic vegetation). |
| | Comparison spatial size and habitat quality to surrounding areas, comparison of receptor habitat preferences to subject property habitat conditions and habitat quality. | | | |
| | Outcomes of the habitat assessment. | | | |

Notes:

1. Weights are High, Medium, Low or Negligible
2. Weights are High, Medium, or Low

5.7 Risk Management

No SSTLs are provided for ROC evaluated in the ERA. The planned risk management/remediation for the study boundary based on the HHRA outcomes (**Section 4.6**), will also reduce lead concentrations (and also strontium concentrations since areas with high lead often also had high strontium concentrations) in soil that ecological receptors would be exposures to, and hence would also reduce potential risks.

Some miscellaneous drums and debris were noted by AEC 1 and south of AEC 12. While not a chemical exposure issue, these drums and debris could pose a physical risk to wildlife in the area. It would be prudent to remove the visible debris and drums.

While the snapping turtle could potentially be exposed to site-related contamination, their exposures would likely be limited due to the limited suitable habitat and food resources, as well as limited uptake of site-related contaminants from ingesting vegetation as evidenced in the aquatic vegetation analytical data (**Appendix L**). It is also unknown if the snapping turtle even currently occurs within the study boundary as none were observed during field work.

Given the wide-spread exceedances of lead in sediments in Lake Enon, conduct fish tissue sampling (filet and whole body) for metals to confirm fish tissue that may be ingested by humans and / or wildlife are not of concern. In addition, if hunting becomes a relevant exposure pathway in the future, it would be prudent to conduct game mammals and / or bird edible tissue sampling for species that are potentially harvested and consumed.

If there are areas of fine tailings that are an open source of lead which could disperse and mobilize, resulting in increased concentrations within the study area boundary or off-site, these areas should be addressed in the remedial plan.

Climate change impacts that could potentially increase concentrations of lead within the study boundary or off-site (e.g., decreased water in tailings ponds such that tailings become exposed, or increased flooding such that contamination spreads into surrounding areas or lakes) should be addressed in the remedial plan.

5.8 ERA Uncertainty Analysis and Conservative Assumptions

As a result of the scientific investigations, literature reviews, and risk assessment guidance followed in the preparation of this ERA, it is believed that the risk assessment results present a reasonable yet conservative evaluation of the risk to ecological receptors present within the study boundary.

In any risk assessment, the findings are based on available data from the specific study area or site, and the scientific literature, in conjunction with a number of assumptions. Every effort is made to ensure that the assumptions and data adequately represent the conditions under evaluation. However, data are often limited, resulting in uncertainty in the assessment. Where uncertainty exists, assumptions are made, and data are selected so as to err on the conservative side. The major sources of uncertainty, limitations and conservatism associated with the current ERA are qualitatively discussed below.

Receptor Selection. Receptors selected for assessment were either known to be present or can reasonably be expected to be present within the study boundary. These receptors are also known to be reasonably or conservatively representative of other species that may be present within the study boundary and exposed to COPCs. While the selection of receptors of concern (ROCs) is believed to be reasonable and appropriate for this site, there is always some possibility that there are species living on (or possibly extirpated from) the area that may be more sensitive to one or more of the COPCs than those receptors that were evaluated in the ERA.

Utilization of Receptors as Surrogates to Represent Other Organisms. The use of receptors as surrogates typically reflects the availability of receptor parameters that can reliably enable ERA of a given receptor species. The receptors selected are considered to be sensitive, and potentially present within the study boundary, and exposed to the COPCs present within study boundary media via relevant exposure pathways. Therefore, it is reasonable to assume that ERA outcomes for surrogate receptors can be extrapolated to similar species that were not or could not be directly assessed in the ERA.

Species at Risk. Species at risk (SAR) have been reported to be seen within the property or could potentially occur within the study boundary. Those species having the potential for exposures from the site were considered (i.e., snapping turtle). There is high uncertainty associated with the potential toxicity of the COPCs to reptilian SAR given the paucity of surface water and sediment-based toxicity data for these species. In addition, there are no reptilian-based guidelines so no COPC screening could be conducted for this species. Given this, potential for exposure (and hence potential for risk) was considered based on site and lifestyle characteristics of the snapping turtle.

Receptor-Specific Toxicity Data. For most COPCs and receptors, toxicity data are available in some form. However, it is important to note that toxicity data are not necessarily available for the particular receptor species under consideration. Also, available toxicity values are not necessarily relevant to the endpoints of greatest interest in an ERA (i.e., growth, reproduction, mortality). As a result, there is uncertainty associated with the extrapolations that may be used to translate toxicity data from one species into a TRV for a second species.

The TRVs selected to derive the EHQs were generally based on the US EPA EcoSSLs. The US EPA (2005d) states that the EcoSSLs are, “intentionally conservative in order to provide confidence that contaminants which could present an unacceptable risk are not screened out early in the ERA process”. The selected

TRVs were based on no-effect concentrations (e.g., geomean of NOAELs; highest NOAEL value lower than the lowest bounded LOAEL value for effects on reproduction, growth, or survival) for the initial TRVs and effects-based concentrations for the secondary TRVs.

Exposure Point Concentrations. The ERA utilized deterministic (or point estimate) exposure analysis techniques. This approach 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 (or other media) contaminant concentrations. This was the case in the current ERA (i.e., UCLM95 soil concentrations of the COPCs were evaluated as the exposure point concentrations (EPCs) for the exposure pathways of soil ingestion). The UCLM95 soil concentration was also used to estimate COPC uptake into vegetation and soil invertebrates and small mammal prey.

Data Limitations. Surface soil, sediment and surface water are considered the most likely media to be impacted from current and historical mine activities. The study boundary is considered to be adequately characterized for soil, surface water and sediments in most areas. Some limited sampling was conducted in the on-site water courses. Near shore sediments in Lake Enon were not collected as the evaluation of these data were outside the scope of the HHERA as per build NS. The collection of additional data in these areas would reduce uncertainty of potential risks to receptors exposed via these media.

The highest EHQ value was for lead in the dark-eyed junco. The junco was assumed to eat 60% soil invertebrates and 40% terrestrial vegetation, and the soil invertebrate and vegetation concentrations were based on the UCML95 soil concentration using standard exposure equations. This would represent a very conservative assumption given the home range of the junco and the habitat on the site. For example, the junco would not feed in the wetland areas of the site given the lack of food resources for them there. In addition, based on the limited amount of vegetation analytical data, which was collected in contaminated areas of Wetland #3 that were recommended for risk management in the HHRA, the estimated exposures via vegetation ingestion for the junco and other receptors have likely been substantially overestimated. Lead concentrations in stem and leaf samples ranged from 0.8 mg/kg (sample 24-WL3-VEG1) to 37 mg/kg (sample 24-WL3-VEG5) with a mean concentration of 7.6 mg/kg. Five of the six analytical results for lead in stems and leaves collected from within Wetland #3 were below the estimated vegetation tissue concentration used in the food chain modeling for the ERA (i.e., 4.4 mg/kg). Therefore, outcomes of the ERA modelling are considered to be reasonably accurate and protective of the ecological receptors that were assessed, given consideration of the Wetland #3 data.

Other Stressors. The current ERA only assesses chemical stressors present in the study boundary soil, sediments, surface water and groundwater. While other common ecological stressors such as predation, disease, habitat loss/fragmentation, competition, limited food resources etc. are likely having some influence on the resident biota within the study boundary, these stressors were not evaluated in either a quantitative or qualitative manner, with the exception of the evaluation of terrestrial vegetation and soil invertebrates (where physical impacts to soil from tailings and rock piles were considered). 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 environmental media.

Chemical Speciation. The environmental fate and behaviour, and toxicity of a number of inorganic contaminants (such as metals) depends to a large extent upon their chemical form. Oral TRVs, however, are typically based on chemical forms in toxicity studies that have high bioavailability. In the ERA, it was assumed that 100% of each ingested COPC is absorbed from ingested food.

Chemical Exposures. Chemical exposures for terrestrial vegetation, soil invertebrates, and freshwater aquatic life were assumed to equal measured soil, surface water and sediment concentrations (as applicable). This is considered to be a conservative assumption as there are limits to the amount of any substance that can be taken up into these receptors from soil, water and sediments, which are a function of the solubility or extractability of the substances in these media, which in turn, affects bioavailability. This assumption overestimates the actual exposure potential, as it ignores natural barriers to chemical uptake (i.e., bioavailability considerations), and biochemical transformation processes that may occur within the rhizosphere (root zone) for plants, as well as within cells, tissues and organs, which may reduce the actual amount of a substance that reaches a target site within an organism. Thus, measured environmental media concentrations likely overestimate the amount of a chemical that may actually be taken up by ecological receptors.

Foraging Behaviour. The ERA was conducted in a manner that assumed that the ROCs forage entirely within the impacted areas of the study boundary and that their respective diets consisted entirely of food items (i.e., terrestrial vegetation, soil invertebrates, prey) from the study boundary. While these are typical assumptions for an ERA, they are considered to be unrealistic and conservative.

Wildlife Exposure Factors. Ecological 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 specific to the study boundary, 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, 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/sediment 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, between 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 the current ERA as well, where receptors were assumed to feed primarily on their majority dietary components. The focus on the majority dietary components is an acceptable method used to estimate exposures based on FCSAP (2013b) guidance.

5.9 ERA Conclusions

The conclusions of the ERA are summarized below, with details for each ROC being provided in **Sections 5.6.1. to 5.6.5.** Risk management measures and recommendations to reduce uncertainties in the ERA have been provided in **Section 5.7.**

The overall conclusions from the ERA include:

- The potential for adverse effects to vegetation and soil invertebrates communities as a result of COPCs is within the study boundary is considered to be negligible to low based on the available data, within most areas of the site. In areas where waste rock and tailings have been placed, the likelihood of potential risks is medium to high. The presence of waste rock and tailing and the lack of nutrients in the “soils” in these areas, have most likely impacted vegetation in these areas and its ability to grow back / repopulate;
- There is the potential for individual level effects in the meadow vole, masked shrew, ruffed grouse, and dark-eyed junco, based on the available data and assumptions used in the exposure and risk modelling. The likelihood of potential effects on local populations of small herbivorous mammals with a moderate home range, carnivorous mammals, small herbivorous birds, or insectivorous birds is considered to be low. Given the home and foraging ranges for the vole and shrew would be encompassed by the areas having elevated metals, and based on the elevated effects-based EHQs, there is a medium potential for effects on local populations of herbivorous and insectivorous small mammals with small home ranges (e.g., shrew and vole);
- The potential for adverse effects in pelagic-feeding freshwater aquatic life communities as a result of contamination on the site is considered to be negligible based on the available data;
- For benthic species, exceedances over freshwater aquatic life sediment quality guidelines occurred in most sampling locations. While the contamination has been present for years and there may be some species acclimation / adaptation, this degree to which this has occurred is unknown. The results of the benthic community, aquatic vegetation, and bivalve tissue analysis indicate that although COCs are present, the overall potential for adverse effects to benthic species as a result of site-related contamination is low;
- Based on the available site data and available amphibian toxicity data, the potential risk to amphibians is considered to be low to medium; and

- While the snapping turtle could potentially be exposed to site-related contamination, their exposures would likely be limited due to the limited suitable habitat and food resources. In addition, the aquatic vegetation investigation revealed only marginal differences between COPC concentrations in near-field vegetation relative to far-field vegetation; therefore, potential uptake of site-related COPCs from vegetation ingestion is expected to be limited. It is also unknown if the snapping turtle even currently occurs within the study boundary, none were observed during field work.

Climate Change Considerations

As discussed in **Section 2.1.4**, the site is situated within the maritime boreal climate zone and is currently characterized by variable weather, warm summers, cold winters, and moderate precipitation; however, impacts from climate change may occur throughout Atlantic Canada further into the future. Warmer temperatures brought about by climate change may incur additional impacts to Atlantic Canada in the form of extreme weather events and changing weather patterns. Annual precipitation under a high emissions scenario is projected to increase by 12% (ClimateData.ca, 2023), but the overall pattern of precipitation may change to drier summers and wetter spring/fall leading to more intense rainfall patterns in these parts of the year.

Extreme weather events (heat waves, flooding, drought, storm events) are typically the drivers of change and are likely to increase in frequency and severity due to climate change (Bush and Lemmen, 2019). Changes to climate may cause a change in habitat within the site and could affect the types and quantities of vegetation and/or ecological receptors encountered within the site.

Climate change impacts that could potentially increase concentrations of lead within the study boundary or off-site (e.g., decreased water in tailings ponds such that tailings become exposed, or increased flooding such that contamination spreads into surrounding areas or lakes) should be addressed in the remedial plan.

Summary of HHERA Conclusions

The HHERA considered potential risks to site visitors; vegetation and terrestrial invertebrates; herbivorous, insectivorous, and carnivorous small mammals; herbivorous and insectivorous birds; freshwater aquatic life; amphibians and the snapping turtle, via relevant exposure pathways to the identified contaminants of potential concern (COPCs). A summary of conclusions and recommendations for HHERA are provided below. These conclusions and recommendations based on the available data and assumptions used in the exposure and risk calculations. If site conditions and/or exposure assumptions change, the risk estimates should be re-evaluated to confirm the conclusions and risk management measures for this assessment still apply.

The overall conclusions from the HHRA include:

- The THQ for lead was exceeded for only the toddler receptor (HQ=3.6). Risk estimates for lead for the child, teen and adult receptors were all below the THQ based on the data and exposure assumptions applied;
- All the strontium HQs were below the THQ for all receptors;
- While there is a potential risk to toddlers in relation to lead, the HQ of 3.6 for the toddler is considered an overestimate. This is largely because it is extremely unlikely that toddlers would be present within the study boundary for any significant duration. The study boundary also has no unique features that would attract or encourage toddlers and their caregivers to be spending time in the lead-impacted portions of the study area. The HHRA conservatively assumed that a toddler would be on the site once per week for 6 months of the year. However, the likelihood of this occurring is considered negligible. The site and study boundary considered in the HHRA is not easy to access for the public and has a number of physical hazards such as unfenced waste rock piles, settling and tailings ponds. The study boundary also contains wetlands and dense forests in some areas. Much of the study boundary was difficult to traverse for Dillon field personnel and this would be even more the case for a toddler. It is highly unlikely that the study boundary would be used for recreational purposes, particularly with young children, when there are numerous other areas in the vicinity of the site and around the shores of Lake Enon that would be far more suitable for families engaging in recreational activities;
- **Section 4.6** discusses proposed risk management recommendations for the study boundary. This included the development of a soil SSSL for lead of 2100 mg/kg, which is protective of all assessed human receptors, including the toddler; and
- **Section 4.7** describes various uncertainties and conservative assumptions applied in the HHRA that are collectively believed to result in substantial overestimates of exposure and risk for all human receptors assessed in the HHRA.

The method or remedy by which the soil SSSL for lead will be achieved will be provided under separate cover from this HHERA report. However, it is anticipated that some combination of excavation, capping, and fencing of certain areas would likely occur.

The overall conclusions from the ERA include:

- The potential for adverse effects to vegetation and soil invertebrates' communities as a result of COPCs is within the study boundary is considered to be negligible to low based on the available data, within most areas of the site. In areas where waste rock and tailings have been placed, the likelihood of potential risks is medium to high. The presence of waste rock and tailing and the lack of nutrients in the "soils" in these areas, have most likely impacted vegetation in these areas and its ability to grow back/repopulate;
- There is the potential for individual level effects in the meadow vole, masked shrew, ruffed grouse, and dark-eyed junco, based on the available data and assumptions used in the exposure and risk modelling. The likelihood of potential effects on local populations of small herbivorous mammals with a moderate home range, carnivorous mammals, small herbivorous birds, or insectivorous birds is considered to be low. Given the home and foraging ranges for the vole and shrew would be encompassed by the areas having elevated metals, and based on the elevated effects-based EHQs, there is a medium potential for effects on local populations of herbivorous and insectivorous small mammals with small home ranges (e.g., shrew and vole);
- The potential for adverse effects in pelagic freshwater aquatic life communities as a result of contamination on the site is considered to be negligible based on the available data;
- For benthic species, several chemicals were selected as COCs for freshwater aquatic life based on concentrations in sediment exceeding their respective freshwater aquatic life-based guideline. The evaluation of benthic community indices such as abundance, taxon richness, EPT ratio, HBI, and the Bray-Curtis Index revealed no strong correlation between contamination levels and benthic community characteristics in Lake Enon or the watercourses. These data suggest that other environmental parameters, such as substrate type, predation, connectivity and general ecological variability may play more significant roles in shaping community assemblages than site-related contamination. In addition, the relatively similar sediment COC concentrations in near-field and far-field aquatic vegetation and bivalve tissues, along with the poor observed relationship between sediment samples co-located at vegetation and bivalve sample locations, provides an additional line of evidence to support that the risk to these organisms from site-related sediment contamination is low;
- Based on the available site data and available amphibian toxicity data, the potential risk to amphibians is considered to be low to medium; and
- While the snapping turtle could potentially be exposed to site-related contamination, their exposures would likely be limited due to the limited suitable habitat and food resources. In addition, the aquatic vegetation investigation revealed only marginal differences between COPC concentrations in near-field vegetation relative to far-field vegetation; therefore, potential uptake of site-related COPCs from vegetation ingestion is expected to be limited. It is also

unknown if the snapping turtle even currently occurs within the study boundary as none were observed during field work.

Risk Management

Proposed risk management recommendations, including additional sampling needs, for the site and HHERA study boundary are as follows:

- The recommended soil SSTL for lead is 2100 mg/kg. This SSTL value is protective of all human receptors that were evaluated in the HHRA, including the toddler, under the exposure assumptions applied. The method or remedy by which the soil SSTL for lead will be achieved will be provided under separate cover from this HHERA report. However, it is anticipated that some combination of excavation, capping, and fencing of certain areas would likely occur;
- Some miscellaneous drums and debris were noted by AEC 1 and south of AEC 12. Some of the drums were partially buried and seemed empty. While not a chemical exposure issue, these drums and debris could pose a physical risk to wildlife or someone walking in the area. It would be prudent to remove the visible debris and drums;
- Preventing the potential flow of water from the on-site culvert into Lake Enon;
- Prevent the movement of contaminated sediments from the tailings ponds / water courses into Lake Enon;
- Develop site-specific health and safety plan for site remediation workers, as required;
- If remedial activities result in freshly exposed waste rock / tailings, these areas should be evaluated for sulfide bearing materials for acid rock drainage potential as warranted. This can be done using a field screen;
- Given the wide-spread exceedances of lead in sediments in Lake Enon, conduct fish tissue sampling (filet and whole body) for metals to confirm fish tissue that may be ingested by humans and / or wildlife are not of concern. In addition, if hunting becomes a relevant exposure pathway in the future, it would be prudent to conduct game mammal and / or bird edible tissue sampling for species that are harvested and consumed.
- If there are areas of fine tailings that are an open source of lead which could disperse and mobilize, resulting in increased concentrations within the study area boundary or off-site, these areas should be addressed in the remedial plan; and
- Climate change impacts that could potentially increase concentrations of lead within the study boundary or off-site (e.g., decreased water in tailings ponds such that tailings become exposed, or increased flooding such that contamination spreads into surrounding areas or lakes) should be addressed in the remedial plan.

9.0

Limitations and Closure

This report was prepared exclusively for the purposes, project and site location outlined in the report. The report is based on information provided to, or obtained by Dillon as indicated in the report, and applies solely to site conditions existing at the time of the site investigation. 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. 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, and that the levels of contamination or hazardous materials may vary across the site. 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 Build Nova Scotia. The material in the report reflects Dillon's best judgment in light of the information available to Dillon at the time of preparation. Any use which a third party (e.g., a party other than Build Nova Scotia) 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.

Respectfully submitted,

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*Human Health and Ecological Risk Assessment and Ancillary Assessment
Activities: Lake Enon Former Mill Site, Enon, Nova Scotia (FINAL)*

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