Project No. CBD-05-03
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Wildlife Screening Level Risk Assessment for the Meadowbank Gold Project

Prepared for:

Cumberland Resources

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Draft

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USE AND LIMITATIONS OF THE REPORT

This report has been prepared by Azimuth Consulting Group Inc. (Azimuth), based to a large degree on environmental impact assessments conducted by others on behalf of Cumberland Resources Ltd. (Cumberland). The extent to which previous investigations was relied on is detailed in the report.

This report is intended to provide information to Cumberland to assist it in making business decisions. Azimuth is not party to the various considerations underlying the business decisions, and does not make recommendations regarding such business decisions. In providing this report, Azimuth accepts no liability or responsibility in respect of the site described in this report or for any business decisions relating to the site, including decisions in respect of the purchase, sale or investment in the site.

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The findings, conclusions and recommendations in this report reflect our best professional judgment and have been developed in a manner consistent with the level of skill normally exercised by environmental professionals currently practicing under similar conditions in the area. The findings contained in this report are based, in part, upon information provided by others and are valid only as of the date of this report. Azimuth has assumed the data or other information provided by others is factual and accurate. If any of the information is inaccurate, site conditions change, new information is discovered, and/or unexpected site conditions are encountered in future work, then modifications by Azimuth to the findings, conclusions and recommendations of this report may be necessary.

This report pertains to a specific site and a specific scope of work. It is not applicable to any other sites, nor should it be relied upon for types of development or remediation other than those to which it refers. Any variation from the site, remediation or proposed development may necessitate a supplementary investigation and assessment.

Risk assessment is a process that manages uncertainty and probability of adverse effects and potential risks of multiple stressors on complex ecological communities. Implicit in this statement is that, by its nature, all information and data used to make decisions are subject to interpretation. Azimuth has used its best professional judgment to make decisions, predictions, and offer opinions about potential risks.

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ACKNOWLEDGEMENTS

This report was prepared by Randy Baker and Alena Fikart of Azimuth Consulting Group Inc. (Azimuth), with contributions from Martin Gebauer of Gebauer & Associates Ltd. Patrick Allard conducted the overall peer review. Contributions by Ross Wilson with respect to the overall ERA process were much appreciated.

Soil and plant tissue samples were collected by Ryan Vanengen (Azimuth) and Damian Power. Analysis of soil and tissue media was conducted by ALS Environmental Laboratories in Vancouver, BC. Rachel Gould (Azimuth) was responsible for collating baseline soil, water and tissue data.

ACRONYMS AND ABBREVIATIONS

AN	Ammonia Nitrate
Azimuth	Azimuth Consulting Group Ltd.
BAF	Bioaccumulation Factor
BCE	British Columbia Ministry of Environment
BQCMB	Beverly and Qamanirjuaq Caribou Management Board.
CCME	Canadian Council of Ministers of the Environment
COPC	Contaminant of Potential Concern
Cumberland	Cumberland Resources Ltd.
EIA	Environmental Impact Assessment
ELC	Ecological Land Classification
EMS	Environmental Management System
EPA	Environmental Protection Agency
ERA	Ecological Risk Assessment
HQ	Hazard Quotient
IF	Iron Formation
IV	Intermediate Volcanic
LOAEL	Lowest Observed Adverse Effect Level
MMER	Metal Mining Effluent Regulations
LSA	Local Study Area
NOAEL	No Observed Adverse Effect Level
NWT	Northwest Territories
ORNL	Oak Ridge National Laboratory
PAG	Potentially Acid-Generating
PAHs	Polycyclic Aromatic Hydrocarbons
PF	Problem Formulation
PSV	Preliminary Screening Value

ROC	Receptor of Concern
RSA	Regional Study Area
SAP	Sampling and Analysis Plan
SLRA	Screening Level Risk Assessment
TDF	Tailings Disposal Facility
TRV	Toxicity Reference Value
UM	Ultramafic
US	United States
VEC	Valued Ecosystem Component

EXECUTIVE SUMMARY

Cumberland Resources Ltd. (Cumberland) is proposing to develop an open pit gold mine on the Meadowbank Gold property, which is located 70 km north of the Hamlet of Baker Lake in the Kivalliq Region of Nunavut, on Inuit-owned surface lands. The proposed mine has an approximate 10 – 12 year life span, including construction (1 – 2 years), operation (8.5 years), and closure phases (1 – 2 years). Post-closure activities and monitoring of open pits during flooding may take several more years.

Potential impacts to wildlife from the proposed mine development were evaluated in the environmental impact assessment undertaken by Cumberland. While these impacts are expected to be low outside of the immediate mine footprint, potential risks from exposure to contaminants via dietary uptake were not specifically assessed. To ensure that potential contaminant release to the environment is minimized, a series of management systems and mitigation measures will be put in place. Notwithstanding these efforts, there will some point source and non-point source losses of contaminants to the aquatic and terrestrial environments. Primary sources of contamination to water and soil are expected to include dust from waste rock piles, mine tailings, and ore storage facilities, as well as mine effluent during the first few years of operation, prior to *in situ* treatment.

To address potential risks to wildlife from mine-related contaminants, Cumberland committed to conducting a wildlife Screening Level Risk Assessment (SLRA). The approach taken to meet this commitment required a somewhat unique application of risk assessment. Specifically, the SLRA focused on 1) evaluating whether contaminants pose potential risks to wildlife under baseline exposure conditions, and 2) quantify the magnitude of increase in contaminant exposure required to trigger potential concern for wildlife populations once the mine is operational. Preliminary estimates of future (i.e., post-development) contaminant concentrations were then obtained from models developed by other team members. Based on potential future changes in contaminant exposure relative to baseline conditions, implications for potential risks to local wildlife were evaluated.

Key findings and recommendations of the wildlife SLRA are presented below:

- Contaminants of potential concern (COPCs) that may be appreciably released during mine operation were identified as follows. First, metals in mine site rock that may be released through dust dispersion were conservatively screened against baseline soil data and federal soil quality guidelines. Second, predicted metals concentrations in Third Portage Lake were screened against federal water quality guidelines. In addition, COPCs included metals that are either regulated under the federal Metal Mining Effluent Regulations or that are known to be of general public concern in the Arctic. Overall, 18 metals were included in the list of

COPCs for the wildlife SLRA. It was assumed that exposure to other contaminants (e.g., petroleum hydrocarbons, cyanide, dioxins) will be negligible based on the implementation of effective source control measures and management plans.

- With the exception of chromium, negligible risks were predicted for all COPCs under baseline exposure conditions. For chromium, potential risks to songbird populations (represented by the Lapland longspur) were considered improbable, but could not be completely ruled out. Monitoring of COPC concentrations in flying insects (i.e., the primary medium driving exposure for these birds) as part of the environmental health monitoring program proposed for the mine (Terrestrial Ecosystem Management Plan, 2005) would address this source of uncertainty.
- Since COPC concentrations in water and soil are not expected to change significantly once the mine is operational, potential risks to wildlife are not expected to increase relative to baseline.
- The environmental health monitoring program for the mine should target the following environmental media: soil, water, plants (e.g., sedges, lichens, berries) and flying insects.
- Preliminary screening values (PSVs) should be developed to support the environmental health monitoring program. These PSVs would serve to guide decision-making by identifying concentrations below which no unacceptable risks would be anticipated. On the other hand, exceedances of the PSVs would lead to further analysis and/or adaptive management. For instance, in the case of chromium, studies of local songbird populations may be required once the mine is operational, if concentrations of this metal in insects exceed baseline estimates.

1. INTRODUCTION

1.1. Background

Cumberland Resources Ltd. (Cumberland) is proposing to develop an open pit gold mine on the Meadowbank Gold property, which is located 70 km north of the Hamlet of Baker Lake in the Kivalliq Region of Nunavut, on Inuit-owned surface lands (**Figure 1**). The proposed mine has an approximate 10 – 12 year life span, including construction (1 – 2 years), operation (8.5 years), and closure phases (1 – 2 years). Post-closure activities and monitoring of open pits during flooding may take several more years.

Potential impacts to wildlife from the proposed mine development were evaluated in the environmental impact assessment (EIA) undertaken by Cumberland (see Terrestrial Ecosystem Impact Assessment, 2005). While these impacts are expected to be low outside of the immediate mine footprint, potential risks from exposure to contaminants via dietary uptake were not specifically assessed. To ensure that potential contaminant release to the environment is minimized, a series of management systems and mitigation measures will be put in place (see Physical Ecosystem Impact Assessment, 2005; Aquatic Ecosystem/Fish Habitat Impact Assessment, 2005). Notwithstanding these efforts, there will some point source and non-point source losses of contaminants to the aquatic and terrestrial environments. Primary sources of contamination are expected to include dust from waste rock piles, mine tailings, and ore storage facilities, as well as mine effluent during the first few years of operation, prior to *in situ* treatment.

To address potential risks to wildlife from mine-related contaminants, Cumberland committed to conducting a wildlife Screening Level Risk Assessment (SLRA). Azimuth Consulting Group Inc. (Azimuth) was retained to carry out the investigation.

The following introductory sections outline the general framework used to guide the risk assessment process and provide details regarding the approach adopted for undertaking the wildlife SLRA.

1.2. Risk Assessment Framework

Ecological risk assessment (ERA) is a process that evaluates the likelihood that adverse effects to ecological resources may occur or are occurring as a result of exposure to one or more stressors (e.g., toxic chemicals). The general ERA framework used to guide the wildlife SLRA for the Meadowbank Gold Project represents an amalgamation of several frameworks used in Canada and the United States (US) (Environment Canada, 1994; BCE, 1998; US EPA, 1992, 1998). Key aspects are as follows (**Figure 2**):

- *Problem Formulation* – The first step in an ERA. The objectives are to understand the nature of contamination at the site (e.g., history of the site; identification of contaminants of potential concern [COPCs] and their sources), determine what receptors of concern (ROCs) may be present, what pathways (i.e., exposure routes from COPCs to ROCs) are likely to be relevant, identify protection goals, and develop a conceptual model for the site (i.e., graphical or written description summarizing key information regarding the site from a risk perspective).
- *Analysis Phase* – This phase consists of two components: exposure assessment (i.e., estimation of contaminant exposure levels for each receptor) and effects assessment (i.e., determination of toxicity reference values [TRVs]). Information is either gathered directly (e.g., surface soil chemistry) or estimated using conservative assumptions (e.g., soil to tissue bioaccumulation models).
- *Risk Characterization* – This phase integrates the results of the two components of the analysis phase. Risks are typically estimated by evaluating the relationship between exposure levels and the TRVs. Depending on the outcome of the risk prediction and/or the number and conservatism of assumptions required to complete the analysis phase, a decision is made as to whether there is too much uncertainty to make a management decision based on the results.
- *Communication and Risk Management* – Some aspects of ERA (problem formulation and risk characterization/management) are based on a combination of science and human values. Accurately reflecting the latter is critical to the success of the risk assessment process, and is achieved through interactions with the proponent, environmental regulators and the public (usually via key stakeholder groups). This process starts at the problem formulation stage (e.g., identifying the ROCs and agreed protection goals) and continues throughout the risk assessment.
- *Information Refinement* – ERA is an iterative process. Initial stages cast a broad net and often rely on conservative assumptions to screen a wide range of ROC/COPC combinations. The conservatism results in a high degree of confidence in "negligible risk" predictions, but often leads to the over-prediction of risks for ROC/COPC combinations not screened out of the ERA. The high degree of uncertainty associated with those "potential risk" predictions usually means that they would not be used to support management decisions regarding a site. Rather, additional data are collected to refine the risk predictions with better information. These iterations are continued until the uncertainty associated with each risk prediction has been reduced to an acceptable level.

Technical guidance considered in preparing the SLRA includes the following documents:

- Guidance and Checklist for Tier 1 Ecological Risk Assessment of Contaminated Sites in BC (BCE, 1998)

- A Framework for Ecological Risk Assessment: General Guidance (CCME, 1996)
- A Framework for Ecological Risk Assessment at Contaminated Sites in Canada: Review and Recommendations (Environment Canada, 1994)
- Guidelines for Ecological Risk Assessment (US EPA, 1998)
- A Framework for Ecological Risk Assessment (US EPA, 1992)

1.3. General Approach for this SLRA

Most applications of ERA involve an evaluation of historical contamination to support risk management of a property (i.e., available COPC exposure data are used to predict risks to ROCs – depending on findings, contamination can then be managed on site or actively remediated). However, the wildlife SLRA for the Meadowbank Gold Project is being conducted within the context of an EIA and, therefore, requires a somewhat unique application of risk assessment. Specifically, potential risks to wildlife need to be assessed under a future (i.e., post-development) exposure scenario without having the benefit of relying on predicted future COPC concentrations. This means that the post-development portion of the SLRA needs to be conducted "backwards" – acceptable environmental concentrations need to be calculated so that the following question can be answered: "By how much would current baseline exposure have to change to result in unacceptable risks to wildlife?" Our approach to address this question involves two tasks:

1. For substances that may be released during mine operation, conduct an SLRA of baseline (i.e., pre-development) conditions and quantify the magnitude of increase in contaminant exposure (i.e., exposure buffers for all ROC/COPC combinations) required to trigger potential concerns for wildlife populations. The baseline SLRA will also serve as a basis for the back-calculation of acceptable media concentrations (see below).
2. Develop preliminary screening values (PSVs) for various environmental media (e.g., soil, water, tissue) which can be integrated into the post-development environmental health monitoring program for the mine. It is stressed that development of acceptable concentrations is inherently a very conservative approach when multi-media exposures need to be considered (even more so than SLRA) since each of the various media need to be assigned an allotted amount of acceptable risks. If future (i.e., post-development) environmental monitoring indicates that concentrations are below PSVs, no unacceptable risks would be anticipated. On the other hand, if post-development monitoring indicates that concentrations are above PSVs, further analysis would likely be recommended prior to reaching firm conclusions (due to the inherent conservative nature of the SLRA).

The scope of work for the present document focuses on the SLRA of baseline conditions (Task 1) with the following components (see risk assessment framework in **Section 1.2**):

- Problem formulation:
  - Identify potential future sources of contamination at the site and screen COPCs.
  - Identify wildlife ROCs and relevant exposure pathways.
  - Present a conceptual model with respect to exposure to potential contamination by receptors.
  - Document baseline COPC exposure concentrations, for the purposes of risk estimation in the SLRA (see below).
- Analysis phase and risk characterization:
  - Develop baseline (i.e., pre-development) exposure estimates for relevant ROC/COPC combinations. Baseline concentrations in soil, water, and tissue will be used to calculate COPC dose rates for various ROCs.
  - Combine calculated exposure estimates with literature-reported TRVs to predict baseline risk estimates from COPCs to each ROC.
  - Quantitatively assess the magnitude of change which must occur for baseline risk estimates to ROCs to become of concern (i.e., exposure buffers).
- Commentary on likelihood of increase in COPC exposure under future (i.e., post-development) conditions:
  - Develop preliminary estimates of future concentrations of COPCs in water (Physical Ecosystem Impact Assessment, 2005) and soils based on major contamination sources (e.g., dust from roads and other sources; Air Quality Impact Assessment, 2005) and discuss potential changes from baseline conditions.
- Commentary on likelihood of increase in risks under future (i.e., post-development) conditions.
  - Based on expected changes in exposure under future conditions, discuss implications for potential risks to local wildlife (i.e., determine if baseline exposure buffers will likely be exceeded).

Once stakeholder consultations have been held to discuss the approach and findings of the baseline SLRA, a separate document will be prepared to address the derivation of PSVs (Task 2) for post-development environmental monitoring at the mine.

Note that this document does not attempt to predict risks for catastrophic events such as fuel spill or other such accident. A variety of management plans and strategies for dealing with such events are described in detail within numerous supporting documents to the EIA including MMER (2005), Mine Waste & Water Management Plan (2005), Air Quality & Noise Management Plan (2005), Emergency Response (2005), Hazardous Materials (2005), and Spill Contingency (2005).

1.4. Report Organization

The wildlife SLRA for the Meadowbank Gold Project contains the following key sections and subsections:

Introduction – This section provides information on project background, the general ERA framework, and the approach taken for the SLRA (**Section 1**).

Problem Formulation – The problem formulation provides a brief description of the site (**Section 2.1**), reviews the major elements of the proposed mine development (**Section 2.2**), summarizes baseline conditions (**Section 2.3**) and potential contaminant sources (**Section 2.4**), screens COPCs (**Section 2.5**), identifies ROCs (**Section 2.6**), proposes protections goals as well as assessment and measurement endpoints (**Section 2.7**), and examines relevant exposure pathways that may lead to ROCs being exposed to COPCs (**Section 2.7**).

SLRA – The baseline SLRA is comprised of an exposure assessment, an effects assessment, a risk characterization, and an uncertainty assessment (**Sections 3.1.3 to 3.1.6**). Additionally, **Section 3.2** provides a commentary on the likelihood of potential increase in COPCs under a post-development scenario, whereas **Section 3.3** discusses the associated risks.

2. PROBLEM FORMULATION

2.1. Site Description

The Meadowbank Gold property is situated in a remote, inland area of east central Nunavut, north of the community of Baker Lake (**Figure 1**). This area is characterized by low, rolling hills that are covered predominantly in heath tundra interspersed with lichen-dominated bedrock outcroppings and boulder fields. The surrounding terrain is typically barren-ground sub-arctic that is dominated by many small lakes with indistinct and complex drainage patterns. These headwater lakes have no large streams associated with them, so there is a paucity of stream habitat in the area as well as organisms that typically inhabit streams.

Extensive deposits of glacial till characterize surficial geology. Eskers are present, although uncommon. The majority of the property is underlain by continuous permafrost with an active thaw layer during summer extending to less than one meter in depth. The remainder of the year the ground is frozen solid.

The property is situated in the extreme headwaters of the Quoich River system that drains into Tehek Lake and eventually flows into Chesterfield Inlet and Hudson Bay (**Figure 1**). To the north, the project lakes lie immediately south of the Meadowbank River, that is part of the Back River system that drains to the Arctic Ocean. All of the lakes in the Meadowbank project area are ultra-oligotrophic, nutrient poor, and very low in dissolved solids and dissolved and total metals concentrations.

Due the extreme northern climate and low structural heterogeneity, relatively few species of terrestrial vertebrates are found in the Meadowbank area (i.e., approximately 15 mammalian species, 62 avian species, and no amphibians or reptiles). Ungulates (primarily caribou) and waterfowl (primarily Canada goose) are relatively common migratory species occurring in the Meadowbank area (see **Section 2.5**).

2.2. Proposed Mining Activities

The Meadowbank Gold Project will extract ore from open pits over an 8 – 10 year operational lifespan. The project will be comprised of several phases, including construction, operation, maintenance, reclamation, closure, and monitoring. All construction and operating supplies for the project will be transported by ocean freight to facilities constructed at the Hamlet of Baker Lake. A 115 km all-weather haulage route from Baker Lake to the project area will provide access and re-supply, while on-site mine access roads will connect the open pit areas to site infrastructure. On-site facilities will include a mill, power plant, maintenance facilities, tank farm for fuel storage, water treatment plant, sewage treatment plant, airstrip, and accommodations. **Figure 3** presents a plan view diagram of the proposed mine during development, depicting major components such as tailings disposal area, waste rock facility, dikes, and pits.

There are two main areas where mining will occur, the Portage – Goose deposit and the Vault deposit, 7.5 km northeast of Portage – Goose (**Figure 3**). Water retention dikes will be constructed from mined rock to allow for the mining of ore beneath shallow lakes.

Ore will be hauled by truck from Vault to a common mill at Portage – Goose. Ore processing will involve cyanide leaching, cyanide destruction, and refining to produce doré bars. The combined leach residue slurry will be treated with metabisulphite to detoxify the free cyanide in the tailings stream. Mine process water will be primarily reclaimed from the tailings pond to minimize water withdrawal from the lake. Treated sewage will be discharged to the tailings pond where it will be buried and freeze.

Mined rock will be placed in tailings impoundments and waste rock storage piles. A classification system will be used to identify both potentially acid-generating (PAG) and metal leaching rock; PAG mine rock will be stored in designated areas designed for long-term stability. Runoff of acidic water will be collected in ditches and directed to attenuation ponds

Both facilities (Portage-Goose deposit and the Vault deposit) will have attenuation ponds that collect contact water (i.e., any water that comes in contact with a potential contaminant sources, such as runoff from waste rock piles, ore storage piles or pit inflow water). Effluent will be discharged from these attenuation ponds to the north basin of Third Portage Lake and the other to Wally Lake. Monitoring of effluent from both sources is subject to the Metal Mining Effluent Regulations (MMER).

Recently, Cumberland's Waste and Water Management Plan was revised, resulting in a significant improvement in the management and discharge of contaminated water, especially effluent, at Meadowbank (Mine Waste & Water Management Plan, 2005). Under the revised plan, effluent will be directed to Goose Pit as part of a closed loop system where the mill takes make-up water obtained from the attenuation pond. After Year 5, Goose Pit will be used to manage all site-water including all site-contact water, pit water (from Portage Pit), waste rock storage run-off, and mill-site run-off. Thus, Goose Pit will function as the repository for effluent and no discharge of effluent to Third Portage Lake is forecast.

Cumberland will implement an Environmental Management System (EMS) consisting of three key elements: an integrated environmental management plan, a formal environmental awareness program, and ongoing environmental monitoring plans. The principal of adaptive management is a key component to each of these elements. Upon conclusion of mining activities, Cumberland will fully decommission the mine by removing the mill and ancillary buildings, access roads, including the all-weather access road between Baker Lake and Meadowbank, and by re-contouring disturbed areas and reclaiming vegetation.

2.3. Baseline Conditions

The following section provides a brief review of baseline conditions at the Meadowbank property. Two exposure media were specifically considered: water and soil. Baseline water quality data were obtained from baseline studies conducted between 1996 and 2005 (BAEAR, 2005), whereas baseline soil data were collected during a comprehensive baseline field study in summer 2005 (see methods in **Appendix A** and summary statistics in **Appendix B**; raw analytical data are provided in **Appendix C**). Note that baseline tissue samples were also collected during the 2005 field study. Chemistry data for these samples were incorporated into the exposure assessment phase of the wildlife SLRA (see **Section 3.1.3**).

2.3.1. Water

A total of 51 baseline surface water samples were collected from Third Portage Lake, Second Portage Lake, Tern Lake, Farside Lake, Amarulik Lake, Tehek Lake, the Vault and Wally lakes, and Inuggugayualik Lake, the external reference lake (BAEAR, 2005). A summary of lake water quality data is provided in **Appendix B**. Water samples were screened against CCME surface water guidelines for the protection of freshwater aquatic life. The guideline value for inorganic mercury and the more conservative value for chromium VI were used. In the absence of freshwater aquatic life guidelines, the most conservative guideline value for other land uses (community and agriculture) was used for the screening.

Water quality of the Meadowbank project lakes is very high owing to their headwater nature, lack of anthropogenic influence and ultra-oligotrophic status. Total and dissolved metals concentrations in surface waters from project lakes from multiple stations, seasons, lakes, and years are low and remarkably similar and do not differ geographically within or between lakes or temporally, between seasons and years.

With the exception of cadmium and mercury, all metals for which there are guidelines concentrations were below CCME (2001) water quality guidelines for the protection of aquatic life (BAEAR, 2005). Although cadmium is not detectable, the detection limit was above the extremely low CCME hardness adjusted guideline concentration. This hardness-derived concentration is theoretical and is more than 100 times lower than the concentration that caused minimal toxicity to the most sensitive test organism (CCME, 2002). Thus, although the detection limit for cadmium exceeds the purported guideline value, this result has no relevance. Mercury detection limits (50 ng/L; i.e., parts per trillion) were above the ultra-low CCME guidelines (26 ng/L). In pristine Arctic headwater lakes mercury concentration is typically less than 1 ng/L.

Other metals and metalloids that were measured, but do not have CCME guideline values include barium, beryllium, boron, calcium, cobalt, iron, lithium, magnesium, manganese, molybdenum, potassium, selenium, silver, sodium thallium, tin, uranium, and vanadium.

2.3.2. Soil

To characterize baseline soil chemistry, 50 surface soil samples were collected from the local study area (LSA) in summer 2005 and analyzed for total metals (see methods in **Appendix A** and data in **Appendix B;** analytical laboratory reports are provided in **Appendix C**).

Concentrations of 26 metals were measured in Meadowbank soils and screened against soil quality guidelines for the protection of environmental and human health from the Canadian Council of Ministers of the Environment (CCME; CCME 1999). The most conservative guideline values from the agricultural, residential/parkland, commercial and industrial land use scenarios were used for screening. Baseline soil concentrations of arsenic, chromium and nickel exceeded the CCME soil quality guidelines for the most conservative land use. Arsenic, chromium and nickel exceeded guidelines in ten, seventeen, and nine samples, respectively. Soil pH was below the CCME guideline of 6 to 8 in 37 soil samples.

2.4. Potential Contaminant Sources

Despite the implementation of a series of management systems and mitigation measures during mine life, small but uncontrollable point source and non-point source losses of contaminants will likely occur. This section summarizes major mine site facilities and reviews their overall contribution to the release of contaminants in water and soil.

Open pits – The Meadowbank Gold Project involves mining of approximately 20 million tonnes (Mt) of ore, which will produce approximately 160 Mt of waste rock. There are three major pits – Portage Pit, Goose Island and Vault pits. Development of these pits takes place at different times during the 8 – 10 year life of the mine. Dewatering of the impounded lake areas to create the pits will not affect the terrestrial environment.

Rock storage facilities – There are two rock storage facilities (**Figure 3**): North Portage and Vault. The Portage rock storage facility, which will hold 97 Mt, begins during the construction phase. The 54 Mt planned for the Vault facility will be placed during the operational phase. Rock stored at these facilities consists of rock that does not contain gold and may consist of a mixture of intermediate volcanic (IV), ultramafic (UM), and iron formation (IF) rock types and will contain a wide variety of grain size materials, but will primarily be larger than gravel size. Dust will be blown from the rock piles during dumping. Both rock piles will be capped with non-metal leaching rock at closure. It is expected that permafrost will rapidly creep into the piles and freeze the entire structure below the cap.

Borrow pits and quarries – Borrow pits and quarries are all within the ultimate pit and waste rock pile footprints.

Tailings Disposal Facility (TDF) – An estimated 21.9 Mt of tailings occupying 15 Mm^3 will be deposited in a portion of the dewatered arm of Second Portage Lake. This material will be quite fine and will be subject to erosion by wind. Factors influencing the magnitude of dust generation include moisture content of the tailings, rate of drying, wind speed and direction, precipitation (rain and snow), season (tailings will rapidly freeze during winter) and timing of deposition (e.g., early in mine life the tailings will be deep in the pit and less exposed to wind than later in mine life). There will also be progressive capping of tailings to minimize loss of fine materials. Again, permafrost is expected to creep upwards and freeze the tailings pile.

It is difficult to predict how much tailings will be dispersed to the LSA by the wind. Nevertheless, despite mitigation measures and given the small grain size of tailings material, the TPF represents a potentially significant source of metals to the surrounding soils via dust.

Roads, airstrip and related traffic – A network of haul roads will connect the ore bodies to the rock storage facilities and the plant site. A pioneer airstrip 764 m long will be provided initially, due to a shortage of available materials. The airstrip will be located on the peninsula separating Second Portage Lake and Third Portage Lake to the immediate north of the plant site. The airstrip and roads will be elevated to reduce problems with snow drifting, and a compactor and grader will be used to maintain the surface year round. Dust suppressants will be applied to the airstrip and roads.

Generic activities such as road dust are considered as part of the road infrastructure on-site, such as the Vault road. The Vault road connects the Vault Deposit to the mill facility and is 7.5 km in length. This road will be used frequently as it will be the main route for transportation of ore to the mill for processing.

Ditches and contact water diversion structures – Ditches are assessed during all temporal phases of the project. Ditches along roads, the airstrip, and those needed to manage contact water are not differentiated. Contact water is defined as any water that may have been physically or chemically affected by mining activities.

Effluent discharge –The impact of effluent discharge on water quality is evaluated during the first five years of mine life (during operations, under MMER; see monitoring MMER plan, 2005). After that point, effluent will be re-directed to Goose Island pit.

Water will be treated *in-situ* if necessary (see **Section 2.2**). Discharge of effluent beyond year 5 of mine operations is not forecast.

Non-contact diversion facilities – Non-contact water is limited to runoff originating from areas unaffected by mining activity and that does not come into contact with developed areas. Non-contact water will be intercepted and directed away from developed areas by means of natural or man-made diversion channels and allowed to flow to neighbouring lakes untreated.

Diesel generating plant, mine plant and associated facilities – Development of the plant site area will require clearing of vegetation and soils to construct the main facilities for ore processing. Three 5 MW diesel plants will provide power to the site. Most pollutants are generated from internal combustion engines and emitted through exhaust. Primary pollutants are nitrogen oxides, hydrocarbons, carbon monoxide and small particulate matter, such as soot and polyaromatic hydrocarbons (PAHs). The Air Quality Impact Assessment (2005) calculated emission rates of PAHs and determined that concentrations were "very low" and did not warrant dispersion modeling.

Crushing and milling of rock for the processing plant will take place under wet conditions and no significant particulate matter emissions are predicted for this activity (Air Quality Impact Assessment, 2005). Potential dry particulate losses to the atmosphere may occur from primary crushing, the ore stockpile, pebble crushing and furnace. Dust control equipment will be an integral component of the mill to minimize loss of particulates especially during primary and pebble crushing (see Air Quality Impact Assessment, 2005 for details). Note that there are stringent requirements to minimize loss of particulates to protect worker health. All of these guidelines will be met.

Mobile sources – In terms of emissions, the worst-case scenario is haulage of ore along the 7.5 km Vault road to the processing plant (Air Quality Impact Assessment, 2005). It is estimated that four trips by each of the 14 vehicles will be made per nine hours. Particulate loss is from diesel emission and entrainment of road dust from vehicle tires has been quantified within the Air Quality Impact Assessment (2005).

Overall, the primary sources of contamination are expected to include dust from waste rock piles, mine tailings, and ore storage facilities, as well as mine effluent during the first few years of operation, prior to *in situ* treatment.

2.5. Identification of COPCs

As described in **Section 1.3**, the wildlife SLRA is intended to assess baseline (i.e., pre-development) conditions and quantify the magnitude of increase in contaminant exposure required to trigger potential concerns for wildlife populations once the mine is operational. To support these objectives, the following approach was used to identify COPCs that may be appreciably released during mine operation:

1. Review naturally (mineralogically) elevated metals in soil/rock.
2. Review water quality predictions for Third Portage Lake (Golder, 2005).
3. Include metals regulated under the federal MMER or metals known to be of general public concern in the Arctic.

Metals in soil/rock – Metals in mine site rock will be released into the environment during mining activities, largely through dust dispersion and contact of waste rock with surface water (see **Section 2.4**). Predicted major sources of dust include the Vault and Portage waste rock piles, the TDF, and mine site roads (Air Quality Impact Assessment, 2005).

Table 1 provides a comparison of average metals concentrations in four dust sources to 1) the 90th percentile of baseline soil concentrations (see **Appendix B**), and 2) soil quality guidelines for the protection of environmental and human health (CCME, 1999). All metals identified in mine rock or tailings exceeding either baseline concentrations or CCME guidelines were identified as COPCs for the wildlife SLRA (**Table 1**).

Note that metals concentrations in each of the four dust sources were estimated as follows (see Physical Ecosystem Baseline Report, 2005):

- The average metals concentration in each rock type known to be present at the mine site was determined.
- The final cumulative proportion of each rock type in each of the four dust sources was calculated. Rock chemistry data was obtained from static testing of pit rock and tailings. The chemistry of dust particles was assumed to be the same as that of the parent material (rock or tailings).

Note that the chemical digestion process that is normally applied to soils (e.g., baseline soils; see **Appendix C**) employs a two-acid *aqua regia* digest to liberate what are conservatively assumed to be "bioavailable metals". Conversely, chemical analysis of the four rock types used a much more aggressive four-acid digest that reveals a more complete picture of actual rock geochemistry. This approach liberates much higher concentrations of all metals than the *aqua regia* process does. Consequently, the COPCs screening process can be considered very conservative.

Water quality predictions - Golder Associates Ltd. (Golder) modeled water chemistry changes in Third and Second Portage Lake during each year of mine development, incorporating contributions from effluent and leaching from dike rock using conservative assumptions (e.g., worst-case chemistry, no loss to sediment, accelerated leach rates from lab data) (see Water Quality Predictions, 2005). **Table 2** presents modeled worst-case predicted water quality results for Third Portage Lake. The chemistry of Second Portage Lake, not shown here, is predicted to be similar. Model results indicate that cadmium was predicted to exceed the CCME (2001) water quality guideline for aquatic life in Third Portage and Second Portage lakes. Manganese was predicted to exceed the aesthetic objective for drinking water.

Note, also, that the predicted cadmium guideline is exceeded despite the ultra-low laboratory detection limit (<0.00005 mg/L) used here. Manganese is conservatively predicted to exceed the aesthetic drinking water guideline (i.e., no toxicity is predicted), primarily because of leaching of manganese from dike material placed in the lake. However, results from field cells on site suggested that loading from dikes based on laboratory tests was much less than was used in the model (see Water Quality Predictions, 2005).

Despite the extremely low likelihood of lake wide toxicity from manganese and cadmium, both metals were included as COPCs in the wildlife SLRA.

MMER and other metals – The following five metals, which are regulated under MMER, were included as COPCs in this SLRA: arsenic, copper, lead, nickel and zinc. Mercury, traditionally a chemical of general public concern in the Arctic, was also included as a COPC.

In summary, the following COPCs were included in the wildlife SLRA:

- Antimony
- Arsenic
- Barium
- Beryllium
- Cadmium
- Chromium
- Cobalt
- Copper
- Lead
- Manganese
- Mercury
- Molybdenum
- Nickel
- Selenium
- Strontium
- Thallium
- Vanadium
- Zinc

Potential contamination from the following sources was not addressed in the SLRA: ammonium nitrate/explosives storage and emulsion plant, mine plant and associated facilities (e.g., fuel islands and storage, process chemicals), and sewage and waste disposal. Chemicals associated with these sources include petroleum hydrocarbons (e.g., fuel islands and storage), process chemicals (e.g., cyanide and cyanide species, metabisulphite), dioxins (waste incinerator), nitrates and ammonia (e.g., rock blasting, sewage disposal, cyanide degradation), and PAHs (e.g., waste incineration, fuel islands and storage).

While these chemicals were not specifically included in the wildlife SLRA, they will be addressed through best management practices, appropriate emergency response plans, spill management plans, and monitoring against background concentrations to trigger further risk-based investigations and/or remediation (see recommendations in **Section 4)**. Implicit in this statement is that future exposure to other COPCs such as petroleum hydrocarbons was assumed to be negligible based on the implementation of effective source control measures and management plans.

2.6. Receptors of Concern (ROCs)

2.6.1. Background

Cumberland conducted extensive baseline surveys of vegetation and wildlife in the Meadowbank area. As part of the EIA conducted for the project, wildlife Valued Ecosystem Components (VECs) were determined based on discussions with stakeholders, public meetings, traditional knowledge, and the experience of other mines in the north. The seven VECs identified through this process include (Baseline Terrestrial Ecosystem, 2005):

- Vegetation (wildlife habitat)
- Ungulates
- Predatory mammals
- Small mammals
- Raptors
- Waterfowl
- Other breeding birds

Of these VECs, four were considered for the SLRA: ungulates, small mammals, waterfowl, and other breeding birds.

Ungulates (primarily caribou), and waterfowl (primarily Canada goose) are relatively common species occurring in the Meadowbank area. As such, they have the potential for exposure to contaminated water, soils, and vegetation, and are used as country foods by Baker Lake Inuit. Because of small home ranges and the greater possibility of localized contaminant-related impacts, small mammals (northern red-backed vole – a common local resident) and other breeding birds (Lapland longspur – the most common breeding songbird species), were also considered in the SLRA.

2.6.2. Ecological Description of Area

The Meadowbank Gold property is situated in an area characterized by low, rolling hills that are covered predominantly in heath tundra interspersed with lichen-dominated bedrock outcroppings and boulder fields. Lakes and ponds are numerous, and are typically clear and nutrient-poor.

An Ecological Land Classification (ELC) of the Meadowbank Regional Study Area (RSA) was developed based on satellite imagery (Baseline Terrestrial Ecosystem, 2005). For wildlife discussion purposes, the 33 identified units were clumped into 10 simplified units: water, sedge, birch and riparian shrub, heath tundra, lichen, lichen-rock, ridge crest/esker/avens, rock & boulder, disturbed, and residual. Within the 5,100 km^2 RSA, the units with the greatest aerial cover were heath tundra (23%), water (19%), lichen (14%), birch & riparian Shrub (13%), sedge (9%), and rock & boulder (9%). Within the 194 km^2 Mine Site LSA, the units with the greatest aerial cover were water (31%), sedge (20%), rock & boulder (15%), lichen-rock (10%), and birch & riparian shrub (9%).

Due the extreme northern climate and low structural heterogeneity, relatively few terrestrial vertebrates are found in the Meadowbank area: approximately 15 mammalian species, 62 avian species, and no amphibians or reptiles. Some species, such as barren-ground caribou, are of very high value to people in Baker Lake, with most Inuit still depending heavily on caribou for food. Muskox are increasing in number in the region but are generally not hunted for food by the Inuit. Large mammalian predators, including wolf, wolverine, and grizzly bear, occur at low densities throughout the region. Small mammals are often abundant and include lemmings and voles. Waterfowl are widely distributed, with Canada goose and long-tailed duck being the most common species. Snow geese are common during the migratory period, but do not nest. Although few songbird species nest in the area, species such as Lapland longspur and horned lark are very common.

2.6.3. Ungulates (Caribou)

2.6.3.1. Biology

In the winter and spring, caribou on the tundra seek areas such as ridge tops and high points of land where snow is relatively shallow (BQCMB, 1999). Later in the season, once green-up has begun, meadows are also used (BQCMB, 1999). Calving occurs in distinct traditional calving areas, none of which are in close proximity to the Meadowbank project. Migration to winter range and rutting occurs in the fall.

Significant inter-annual changes in distribution and abundance in the Meadowbank area occur with densities in some years not necessarily being a good predictor of densities in subsequent years. In general, low numbers occur in the summer months and higher numbers occur in the fall and during the winter.

Average body weight of an adult caribou is 75-125 kg (Dauphine, 1976).

2.6.3.2. Diet

Caribou summer diet in the central Canadian Arctic is dominated by willow (Van Egmond and Rowell, 1998), lichens, forbs and graminoids (Boertje, 1984; Barten et al, 2001). The fall diet is varied, consisting of grasses, woody plants, lichens, and mushrooms (Miller, 1976; Russell, 1998). The winter diet is dominated by terrestrial lichens of the *Cladina*, *Cladonia*, and *Cetraria* species (Miller, 1976). Lichens comprise 41 to 61% of caribou winter forage in the Kivalliq region (Fischer et al., 1977; Thompson et al., 1978) and up to 90% of the diet of other caribou populations in Nunavut (Russell et al., 2002). Lichens are poor in protein but high in digestible carbohydrates and become particularly important to caribou in winter, when they are the only forage that is abundantly available. Russell and Martell (1984) summarized caribou diet preferences. Diet was variable and depended upon season: lichens (5-90%), mosses (5-50%), grasses (5-40%), forbs (0-10%), and shrubs (5-40%).

2.6.3.3. Distribution

Caribou are found throughout Nunavut. Radio-collaring data from the governments of Nunavut and Northwest Territories suggest that individuals wintering in the Meadowbank area may originate from any one of several identified herds including the Beverly, Qamanirjuaq, Lorillard, Wager Bay, Boothia Peninsula, and Ahiak (Queen Maud) herds. Although their winter ranges may coincide, these herds are identified by their discrete traditional calving grounds.

Based on patterns of seasonal abundance and distribution observed to date, the Meadowbank area does not represent critical caribou habitat during spring migration, calving, or summer post-calving.

The population of caribou within the Meadowbank RSA is generally at its lowest in early summer when the majority of the herds are on their calving grounds (Russell et al., 2002) and most abundant in winter. In February 2004, caribou densities within the Meadowbank RSA were estimated at 2.1 caribou/km^2 whereas the densities in April 2004 were estimated at 0.4 caribou/km^2. Summer densities are even lower.

2.6.3.4. *Home Range*

Since most caribou in Nunavut undergo long distance migrations, the annual home range of an individual caribou is very large (~10,000 km^2). Radio-collaring data have documented extensive movements by animals of most of the major herds expected to occur in the Meadowbank area.

In winter, caribou are relatively sedentary, moving only about 5 km per day (Russell and Martell, 1984). Nevertheless, since Arctic tundra caribou make frequent and unpredictable winter range shifts (Ferguson et al., 1998; Buckland et al., 2000; Ferguson et al., 2001), the importance of the Meadowbank area as winter range is expected to vary over the long-term.

2.6.4. Waterfowl (Canada Goose)

2.6.4.1. *Biology*

Canada goose generally nests in short vegetation (e.g., sedges, dwarf shrubs, lichens) on islands in small tundra lakes and ponds, or in riparian borders of lakes, ponds, and rivers (Ehrlich et al., 1988; Mowbray et al., 2002). Fidelity to breeding sites is not known, but is believed to be high (MacInnes et al., 1974).

Brood-rearing and moulting habitats are similar to breeding habitat and generally consist of ponds, lakes, and rivers with abundant food resources (Mowbray et al., 2002). Similar habitats are used during migration, although upland heath and grassy fields are also used (Mowbray et al., 2002).

Clutch size is 4-7 but can be up to 10 (Ehrlich et al., 1988). Incubation period is 25-30 days and the fledgling period is 40-73 days (Godfrey, 1986; Ehrlich et al., 1988).

Average body weight of a Canada goose is 2-4 kg (Mowbray et al., 2002).

2.6.4.2. *Diet*

In breeding areas, food consists of shoots, roots and seeds of grasses and sedges, and bulbs, grains and berries of other plants. During migration, particularly in fall, foraging also occurs in upland heath tundra areas where berries from *Vaccinium* and *Empetrum* are important (Godfrey, 1986).

Cadieux et al. (2005) found that in early summer, Canada goose fed primarily on grasses, sedges, and forbs (>65%). Based on Cadieux et al. (2005), the diet of Canada goose at Meadowbank in late summer and during fall migration is expected to consist of sedges (30%), grasses and forbs (20%), and shrubs (50% - mainly berries).

2.6.4.3. Distribution

In Nunavut, Canada goose breeds throughout mainland areas and southern areas of Victoria and Baffin islands. In the Meadowbank area, it occurs at low densities but has been observed breeding on small ponds and lakes. Canada geese are common in migration with many birds stopping to forage in wetland and heath tundra areas.

2.6.4.4. Home Range

Canada geese are highly migratory with birds breeding in Nunavut migrating south to winter in southern Canada, the US, and northern Mexico. On their breeding grounds, Canada geese are closely associated with the wetland or pond that they nest on (1-10 ha).

2.6.5. Songbirds (Lapland Longspur)

2.6.5.1. Biology

Lapland longspurs typically nest on relatively flat ground in wet, hummocky, tundra meadows, and on well-vegetated drier slopes (Hussell and Montgomerie, 2002). In the Yukon, a variety of open tussock tundra habitats in both wet and dry areas were preferred (Alexander et al., 2003). Nests are in a shallow depression of moss, sedge or grass and are usually well concealed under shrubs (Godfrey, 1986; Ehrlich et al., 1988).

In a 1983 study in the Yukon, mean clutch size and mean brood size of 45 nests was five and four, respectively (Dickson et al., 1988). Average clutch size across its range varies from four to six (Godfrey, 1986; Ehrlich et al., 1988). The short period when snow-free tundra is available generally prevents more than one brood from being raised successfully; however, replacement clutches are occasionally laid if the first clutch is lost during laying or early incubation. Incubation period is 11-12 days while fledgling period is not well know but likely ranges between 9-14 days (Ehrlich et al., 1988).

Average body weight of an adult Lapland longspur is 23-33 g (Hussell and Montgomerie, 2002).

2.6.5.2. Diet

The Lapland longspur's diet consists primarily of plant seeds at the beginning and end of the breeding season, while arthropods (mainly dipterans and hymenopterans) and spiders are taken throughout the rest of the season (Ehrlich et al., 1988; Hussell and Montgomerie, 2002). Young are fed 100% insects (Ehrlich et al., 1988).

2.6.5.3. Distribution

Lapland longspurs have a circumpolar distribution breeding in much of Alaska, northern Canada, Greenland, and northern Eurasia (Godfrey, 1986). In Nunavut, they breed in all of the mainland and island areas with the exception of the northern Ellesmere Island and some of the more northern Queen Elizabeth Islands (Godfrey, 1986).

2.6.5.4. Home Range

At Meadowbank, the average number of Lapland longspur pairs per 40 ha ranged from 10 to 20 (maximum of 30), representing an average home range size of approximately 2 to 4 ha (see Terrestrial Ecosystem Baseline Report). In Alaska, Seastedt and MacLean (1979) reported densities averaging about 23 pairs (range 17-37) per 40 ha. Also in Alaska, Bent (1968) reported 14 pairs per 40 ha in low riparian willows and up to 65 pairs per 40 ha in sedge meadows. Tryon and MacLean (1980) reported individuals using 3 to 8 ha for 95% of foraging activity in the breeding season.

2.6.6. Small Mammals (Northern Red-backed Vole)

2.6.6.1. Biology

The northern red-backed vole occurs in all tundra habitats except those that are exclusively sedges and grasses (Martell and Pearson, 1978). Their niche is broad because their morphological, physiological and ecological characteristics can be variable (Whitney, 1976; Douglass, 1984).

The breeding season is from May to August with females having up to three litters. Each litter consists of between 4-9 young after a gestation period of 17-19 days. Average lifespan is 10-12 months with a maximum of 20 months (Anand-Wheeler, 2003).

Red-backed voles exhibit marked population fluctuations (Douglass, 1984; Gilbert and Krebs, 1991). They are active throughout the year.

Average body weight of an adult northern red-backed vole is 20-30 g (Nagorsen, 2005).

2.6.6.2. Diet

Red-backed voles feed primarily on leaves, twigs, buds, and fruit from a wide range of shrubs. They also eat forbs, fungi, lichens, and insects but do not eat mosses. Voles gather and store food in their nests and the stored food comprises most of their diet during the winter months (West, 1977; Bangs, 1984; Zuercher et al., 1999; Anand-Wheeler, 2003).

Based on Nagorsen (2005), the spring and summer diet of northern red-backed vole is likely 90% grasses, sedges and forbs, and 10% mosses. In the fall, diet likely consists of 80% berries, and 20% grasses, sedges, and forbs. In winter, diet may consist of 60% berries, and 40% grasses, sedges, and forbs.

2.6.6.3. Distribution

The northern red-backed vole is found throughout much of northern Canada (Banfield, 1974). In Nunavut, it can be found throughout the Kivalliq region and into parts of the Kitikmeot and Baffin regions (Anand-Wheeler, 2003). Northern red-backed voles have been observed incidentally on several occasions in the Meadowbank area (see Baseline Terrestrial Ecosystem Report).

2.6.6.4. Home Range

Home range tends to decrease (and overlap with others) during winter and increase during warmer months. A female's home range can be from 0.5 to 0.1 hectares in size (Batzli, 1999). In Alaska, Douglass (1984) found that populations demonstrated marked intra-annual fluctuations (3/ha to 37/ha).

2.7. Protection Goals and Endpoints

The protection goals for a risk assessment define the degree of adverse impact considered acceptable for a specific ROC. Rare and endangered ROCs are typically afforded protection at the level of individual organisms, while non-threatened ROC species are protected at the level of populations. Since the four ROCs identified for the wildlife SLRA (**Section 2.6**) represent common species, they will be protected at the population level.

Assessment and measurement endpoints essentially provide progressively more detail regarding the protection goals for ROCs. An assessment endpoint is a formal statement identifying the specific characteristics of receptors that are targeted for protection. Considerations in selecting assessment endpoints include ecological relevance, policy goals and societal values, and susceptibility to the COPCs. The following assessment endpoint was identified for the SLRA:

- Maintenance of healthy populations of caribou, Canada goose, Lapland longspur, and northern red-backed vole not carrying unacceptable body burdens of bioaccumulative substances.

Measurement endpoints are defined as measurable responses to COPCs linked to the valued characteristics chosen as assessment endpoints. Measurement endpoints are the same for each ROC identified for the SLRA:

- Food chain modeling comparing exposure from COPC uptake (via soil, water and food items) to literature-based ecotoxicological benchmarks which reflect COPC doses equivalent to maximum acceptable exposure levels for each ROC.

2.8. Conceptual Site Model

A conceptual model is a graphical representation of our understanding of how COPCs can potentially affect ROCs. One of the important outputs of the conceptual model is the identification of exposure pathways for each ROC.

For the purposes of conducting an SLRA of baseline conditions, the following pathways were considered most relevant:

- Invertebrates (e.g., flying and ground insects) take up COPCs through soil ingestion and direct soil contact.
- Plants (sedges, lichens, berry producers) uptake COPCs from soil via their root system.
- Herbivorous small mammals (e.g., northern red-backed vole) may be exposed to COPCs through ingestion of plants, berries, some insects, water, and incidental ingestion of soil.
- Herbivorous ungulates (e.g., caribou) may be exposed to COPCs through ingestion of plants, water, and incidental ingestion of soil.
- Omnivorous songbirds (e.g., Lapland longspur) may be exposed to COPCs through ingestion of plants, berries, insects, water, and incidental ingestion of soil.
- Omnivorous waterfowl (e.g., Canada goose) may be exposed to COPCs through ingestion of plants, berries, insects, water, and incidental ingestion of soil.

Note that this conceptual model represents current baseline (pre-development) conditions; however, COPCs were identified under a future (post-development) scenario (**Section 2.4**).

3. SCREENING LEVEL RISK ASSESSMENT

3.1. Potential Risks under Current (Pre-development) Conditions

3.1.1. Approach

The purpose of this section is to summarize the approach taken to 1) determine whether COPCs pose potential risks to wildlife under baseline exposure conditions, and 2) quantify the magnitude of increase in contaminant exposure (i.e., exposure buffers for all ROC/COPC combinations) required to trigger potential concern for wildlife populations once the mine is operational. The wildlife SLRA involved two components common to most risk assessments: an exposure assessment and an effects assessment. Results of these components were subsequently combined during a risk characterization stage. Key elements of the SLRA approach are presented below.

Exposure assessment – The exposure assessment relied on food chain modeling to integrate many of the key factors likely to influence the magnitude of risks to individual ROCs:

- COPC concentrations in soil, drinking water, and food items (sedges, lichens, berries, and insects)
- Dietary preferences
- Soil, water, and food ingestion rates
- Dose adjustment factors

Modeling calculations followed guidance from key technical resources (Sample et al., 1996; BCE, 1998). Primary literature was also consulted and incorporated into the modeling approach to ensure that the most ecotoxicologically relevant information was being applied to the site (e.g., soil-to-tissue bioaccumulation factors for particular food items, ROC-specific dietary preferences, and biological characteristics). For instance, most of the input parameters were derived from the Oak Ridge National Laboratory (ORNL) guidance documents (Sample et al., 1996), wildlife literature (**Section 2.6**) and the Wildlife Exposures Handbook (US EPA, 1993). The model was constructed in Microsoft Excel using multiple worksheets to organize and differentiate input parameters, estimated dietary doses, and model outputs.

Effects assessment – The effects assessment primarily involved the identification of literature-based ecotoxicological benchmarks that reflect chemical concentrations or doses that are equivalent to maximum acceptable exposure levels for ecological receptors. These benchmarks are generally referred to as TRVs. For instance, if exposure levels do not exceed a TRV, no unacceptable risks to ROCs would be expected. However, if exposure levels exceed a TRV, it does not imply that unacceptable effects are occurring but rather that additional investigations may be required to reduce uncertainty and refine risk estimates. TRVs (i.e., based on a lowest observable adverse effect level [LOAEL[1]]) were used to protect populations of common wildlife species.

Risk characterization – Results of the exposure and effects assessments were combined during the risk characterization stage to derive HQs that indicate the magnitude of potential ecological risks. HQs were calculated by dividing the modeled exposure (or dose) by the maximum concentration not expected to result in unacceptable toxic effects (i.e., TRV).

The HQ results were interpreted as follows[2]:

- HQ <1 indicate that baseline risks were negligible.
- 1< HQ <10 indicate low to moderate potential for adverse effects under baseline exposure conditions.
- HQ >10 indicate moderate to high potential for adverse effects under baseline exposure conditions.

Using the calculated HQs, exposure buffers (i.e., 1/HQ) were calculated for each ROC/COPC combinations to determine by how much baseline exposure would have to change to result in unacceptable risks to wildlife.

The subsequent sections present the details of the SLRA methods.

3.1.2. Representative Species for the Food Chain Model

In **Section 2.6** of the problem formulation, four specific ROCs were identified for evaluation:

- Caribou
- Canada goose

[1] The LOAEL-based TRVs were based on an EC20 threshold (i.e., dose resulting in a 20% reduction in an endpoint such as reproduction or growth) to ensure that the protection goals were met.

[2] Note that HQs > 1 do not necessarily mean that COPCs are adversely affecting ROCs. There is substantial conservatism in the SLRA that warrants further refinement before any conclusions regarding risks can be made with confidence.

- Lapland longspur
- Northern red-backed vole

These ROCs are relatively common species occurring in the Meadowbank area and, therefore, have the potential for exposure to contaminated water, soils, and vegetation once the mine is operational. Caribou and Canada geese are also used as country foods by Baker Lake Inuit.

3.1.3. Exposure Assessment

The exposure assessment presents the data sources as well as the procedures and calculations used to derive the predicted doses of each COPC to the ROCs. Supporting food chain model spreadsheets have been included in **Appendix D**. In general, exposure was calculated using the following model input parameters: soil and food COPC concentrations (measured or modeled), soil and food ingestion rates, dietary preference, and dose adjustment factors (COPC uptake efficiency). Each will be discussed separately in the following sections.

3.1.3.1. COPCs for Inclusion in the Food Chain Model

Based on results of the screening process presented in **Section 2.5**, a total of 18 metals were included in the wildlife food chain model:

- Antimony
- Arsenic
- Barium
- Beryllium
- Cadmium
- Chromium
- Cobalt
- Copper
- Lead
- Manganese
- Mercury
- Molybdenum
- Nickel
- Selenium
- Strontium
- Thallium
- Vanadium
- Zinc

3.1.3.2. Measured Dietary Concentrations

COPC concentrations in soil and plant tissue (i.e., sedges, lichens, and berries) were directly measured during the 2005 baseline field studies (see overview of methods in **Appendix A**).

Soil – Soil concentrations used for food chain modeling were derived from baseline data collected in the Meadowank LSA (**Table 3; Appendix B**). Specifically, individual soil concentrations were used to calculate 95% upper confidence limit of the mean (UCLM) exposure estimates for soils and to predict tissue concentrations for insects (see **Section 3.1.3.3**).

Water – Drinking water concentrations used for food chain modeling were derived from baseline data from Third Portage Lake (**Table 3; Appendix B**). Specifically, individual water concentrations were used to calculate 95% UCLM exposure estimates.

Tissue – Plant concentrations (i.e., sedge, lichen, and berries) used for food chain modeling were derived from baseline data (**Table 3; Appendix B**). Individual tissue concentrations were used to derive 95% UCLM exposure estimates.

Mercury in soil and tissue – The organic forms of mercury (e.g., methyl mercury) are more toxic and more bioaccumulative than the inorganic forms. To provide realistic predictions for risks associated with organic mercury, the total mercury concentrations measured in soil and tissues were partitioned into inorganic and methyl mercury fractions. The inorganic fraction was calculated as the total mercury minus the methyl mercury fraction; the latter was estimated as follows:

- Soil: proportions of methyl mercury in soil are generally very low (i.e., < 1%), especially in cases where elemental mercury is the likely source (Gnamus and Horvat, 1999; Turner pers. comm., 2002). US EPA (1997) reported that much of the methyl mercury in soil likely results from the wet deposition (i.e., contaminant-laden rain) of divalent mercury (Hg[II]), which is the species used by bacteria in the methylation process. Therefore, for modeling purposes, the methyl mercury fraction in soil was conservatively estimated at 1%.
- Water: for modeling purposes, it was assumed that proportions of methyl mercury in water are negligible (1%) compared to inorganic mercury.
- Plants (sedges, lichens, and berries): using data from Moore et al. (1995) and Gnamus and Horvat (1999), the proportion of methyl mercury in plant tissue was estimated at 34%, conservatively based on the 90^{th} percentile of available data cited in these studies.

3.1.3.3. *Estimated Dietary Concentrations*

Insect tissue concentrations were not measured as part of the 2005 baseline field studies. Consequently, they needed to be calculated to support the food chain model for insectivorous species such as the Lapland longspur. Although the latter consumes primarily flying insects, general arthropod tissue concentrations were estimated and used as a conservative surrogate due to lack of applicable information for flying insects. The modeling approach developed by Sample and Arenal (2001) was used to predict COPC accumulation from soils to ground insects. In general, their approach involves the compilation of soil and invertebrate concentration measures. Soil-arthropod bioaccumulation factors (BAFs) and loglinear bioaccumulation models were then calculated similar to other models developed for earthworms (Sample et al. 1999).

Bioaccumulation models for ground insects were available for the following five COPCs:

- Arsenic
- Cadmium
- Copper
- Lead
- Zinc

A BAF of 1 was assumed for all remaining COPCs.

3.1.3.4. *Food Ingestion Rates*

Food ingestion rates were obtained from US EPA (1993), where available. In cases where values could not easily be located, ingestion rates were estimated using allometric equations based on individual feeding guilds (US EPA, 1993). Food intake was first calculated on a dry weight basis and converted to wet weight using approximated food moisture contents (see above literature sources).

3.1.3.5. *Soil Ingestion Rates*

Soil ingestion rates were primarily calculated following the approach presented in US EPA (1993). Total dry weight food ingestion rates were calculated using an allometric formula presented in the document. This formula uses both feeding guild and body weight data to approximate the total dry food ingestion rate. This rate was then multiplied by the incidental soil ingestion - expressed as a percentage of dry food ingestion. Soil ingestion percentages were compiled for numerous species (e.g., from Beyer et al., 1994); however, some of the representative receptor species chosen for the model were not listed. If this was the case, the most appropriate value was selected based on similarities in feeding behavior. Where substitutions did not appear suitable, alternative sources of soil ingestion were investigated (e.g., ORNL documents such as Sample and Suter [1994] and Efroymson et al. [1997]). Note that in cases where a fraction was stated as being below a 2% value, a conservative approach was taken by adopting the maximum value (i.e., 2%).

3.1.3.6. Dietary Preferences

In addition to the ingestion rate and concentration of COPC in a food item, the calculated dose is a function of the dietary preferences of the ROC. The proportion of a given food type in the diet is multiplied by the measured/estimated concentration giving a measure of how much a single food item contributes to the overall dose.

Dietary preferences, which were primarily obtained from the literature (see **Section 2.6**), are summarized in **Appendix D** for each ROC.

For modeling purposes a number of food item substitutions were made:

Sedges, lichens, and berries – Sedges, lichens, and berries were considered surrogates for all ingestible plant matter. While COPC accumulation may differ among plants, these surrogates represent a range of food items that are likely to constitute a large portion of the diet of most herbivores.

Invertebrates – Ground insects (e.g., beetles, grasshoppers, centipedes) were considered surrogates for all invertebrates due to a lack of bioaccumulation models for estimating COPC concentrations in flying insects. This substitution is considered conservative since ground insects are likely exposed to greater COPC concentrations in soils than flying insects.

3.1.3.7. Dose Adjustment Factors

The dose adjustment factors incorporate information on ROC and COPC characteristics that significantly increases the realism of exposure and risk estimates. Specifically, the dose adjustment factors integrate modifying parameters such as contaminant bioavailability and foraging range of individual receptors into the calculation of the total predicted dose (see **Appendix D**). Primary dose adjustment factors and how they were used in the food chain model are described below.

Territory and foraging range – While this dose adjustment factor can greatly influence exposure estimates when animals have large foraging ranges (e.g., caribou) compared to the size of a particular site, it was not considered in the SLRA of baseline conditions. Rather, it was assumed that all portions of an animal's foraging range were similar in terms of COPC concentrations. This, however, will not be the case once the mine is operational (i.e., the footprint where any increases in COPC exposure may apply will be small compared to foraging ranges) (see **Section 3.3**).

COPC uptake efficiency factors – A fraction of the total contaminant ingested is actually absorbed across the gastro-intestinal tract. Items that are poorly digested typically have lower COPC uptake efficiencies (e.g., soils ingested directly may have a lower bioavailability than contaminants present in tissues). This factor was conservatively assumed to be 1 in the model (i.e., 100% uptake of ingested contaminants).

3.1.4. Effects Assessment

The effects assessment summarizes the potential ecological effects of the COPCs and outlines the methods used to derive avian and mammalian TRVs.

TRVs represent the maximum daily dose (mg chemical/kg wet weight per day) of a contaminant not associated with unacceptable adverse effects. TRVs were derived for each ROC/COPC combination in the food chain model by combining BCE (1998) guidance with supporting information available from ORNL documents (Sample et al., 1996). While contaminant intake toxicological studies have been conducted with most COPCs on a variety of birds and mammals, direct studies are limited for most ROCs. Consequently, extrapolation from tested species is usually required to develop TRVs for each ROC/COPC combination. Generally, avian test species were used to calculate avian TRVs and mammalian test species were used to calculate mammalian TRVs.

When selecting a literature toxicity study to derive a TRV, it is important to ensure that the study was conducted in a manner relevant to its use in the SLRA. Consequently, the following general criteria were used to select appropriate literature toxicity studies for test species:

- The study was preferred if the test species shared the same general phylogeny (i.e., bird or mammal).
- Relevant ecological endpoints (i.e., growth, developmental and reproductive endpoints) were selected over endpoints that were not directly associated with either individual- or population-level effects (e.g., biochemical changes such as enzyme induction were not selected).

- Chronic studies (i.e., >1 year long for mammals; >10 weeks long for birds) were preferred over subchronic studies, which were preferred over acute studies. Any reproduction or early development tests conducted over critical developmental stages (i.e., test initiation immediately or shortly after birth) were also considered chronic studies.
- The most relevant study meeting most or all of the above criteria was then chosen.

Unlike aquatic toxicity studies where many concentrations of a particular contaminant can be tested without significant logistical or financial problems, wildlife toxicity testing requires a scale and duration that typically prevents the simultaneous examination of many test concentrations. Consequently, most available studies test a few concentrations and can only report whether or not adverse effects were observed at a particular concentration. As such, the ideal situation is to have an effects threshold bounded (i.e., at least one test concentration associated with no effects and the next highest concentration associated with an adverse effects). Some studies can only report whether or not a particular concentration was associated with an adverse effect (i.e., effects or no effects concentrations are unbounded).

Where available, two TRVs were derived for each ROC/COPC combination: one based on the no observable adverse effect level (NOAEL; TRV_{NOAEL}) and the other based on the lowest observable adverse effect level LOAEL (TRV_{LOAEL}). The TRV_{NOAEL} corresponds to the highest dose in the selected study that does not result in any unacceptable adverse effects (n.b., many studies report effect concentrations in terms of food concentrations and not dose; reported food concentrations were standardized to dose where required). The TRV_{LOAEL} corresponds to the threshold dose above which unacceptable adverse effect may occur. As mentioned above, literature toxicity studies were not always able to report bounded NOAELs and LOAELs. In the case of an unbounded LOAEL (i.e., where no suitable NOAEL could be derived from literature), the TRV_{NOAEL} was estimated by multiplying the LOAEL by 0.1 (e.g., arsenic for mammals and cobalt for birds and mammals). Note that while this approach is consistent with Sample et al. (1996), Chapman et al. (1998) caution that LOAEL-to-NOAEL extrapolations can be associated with a high degree of uncertainty. In the case of an unbounded NOAEL (i.e., where no suitable LOAEL could be derived from literature), no valid extrapolation technique was available to estimate a TRV_{LOAEL}.

As appropriate, the derivation of both TRV_{NOAEL} and TRV_{LOAEL} values allows the flexibility of providing different protection goals for different ROCs. For example, the TRV_{NOAEL} is conservative but suitable to meet the protection goals for individual organisms (i.e., rare and endangered species). As mentioned in **Sections 2.7 and 3.1.1**, the TRV_{LOAEL} values used in this study should be considered conservative since most are considered to approximately represent an EC20. Accordingly, the TRV_{LOAEL} values were adopted for all common species.

The form of contaminant with the greatest toxic potency was generally chosen in order to calculate a worst-case scenario. Chromium-6 was conservatively chosen for mammals; however, only chromium-3 toxicity data were available for birds.

Once a test species was selected (see below for more details) for a COPC and receptor type (i.e., mammal or bird), then the NOAELs and/or LOAELs for the test species required extrapolation to the ROCs. For mammalian ROCs, the TRVs were allometrically scaled (adjusted for body weight) from a selected toxicity experiment, typically performed with a mouse or rat. For birds, scaling factors for the majority of chemicals evaluated by Mineau et al. (1996) were not significantly different than 1. Accordingly, a scaling factor of 1 was considered appropriate for interspecies extrapolations among birds. While the use of scaling factors does have some drawbacks (e.g., limited number and type of chemicals upon which current models are based), this approach is simple to apply and is considered by Suter et al. (2000) to have a stronger scientific basis than uncertainty factors.

Mammalian or avian test species were not available for either NOAEL or LOAEL-based total mercury TRVs. Mammalian test species were not available for LOAEL-based TRVs for inorganic mercury and beryllium. Avian test species were not available for LOAEL-based TRVS for beryllium, manganese, strontium, and vanadium. In cases where mammalian or avian test species were not available, no TRVs were derived.

Table 4 presents the TRVs used for the assessment of potential risks to mammals and birds. Sample et al. (1996) provided TRVs for most of the COPCs; however, values for antimony, cobalt, and thallium were obtained from other sources:

- Antimony for mammals (Rossi et al., 1987; Dieter et al., 1991, cited in Lynch et al., 1999)
- Antimony for birds (extrapolated from antimony TRV for mammals)
- Cobalt for mammals (Chetty et al., 1979; Szakmary et al., 2001
- Cobalt for birds (Van Vleet, 1982)
- Thallium for birds (Ueberschar et al., 1986)

3.1.5. Risk Characterization

The purpose of this section is to 1) summarize potential risks to wildlife ROCs from exposure to COPCs associated with baseline conditions in the Meadowbank Gold Project area, and 2) document the magnitude of change in COPC exposure required to trigger potential concern for wildlife populations once the mine is operational. A detailed uncertainty assessment conducted for this stage of the SLRA is presented in **Section 3.1.6**.

The overall risk estimates (i.e., HQs) for each ROC/COPC combination are presented in **Table 5.** Key findings are as follows:

Northern red-backed vole – negligible risks (i.e., HQs < 1) for all COPCs.

Caribou – negligible risks (i.e., HQs < 1) for all COPCs.

Lapland longspur – negligible risks (i.e., HQs < 1) for all COPCs, except chromium for which low risks are predicted (HQ of 2.5). These low risks are almost entirely driven by the chromium concentrations in insects, which were estimated (not measured in the field) using BAFs (see **Section 3.1.3.3**).

Canada goose – negligible risks (i.e., HQs < 1) for all COPCs.

Given the conservatism inherent in the model and its assumptions, these results provide a high degree of certainty in predictions of negligible risks. In the case of Lapland longspur and chromium, the HQ of 2.5 does not necessarily imply that this species is adversely affected under baseline exposure conditions. For instance, soil concentrations of this metal already exceed CCME guidelines which suggests naturally elevated levels of chromium (see **Section 2.3.2**). Therefore, it is probable that local wildlife, including songbirds have already acclimated to these exposure conditions. In addition, COPC concentrations in insects (i.e., the primary medium driving exposure to the Lapland longspur) were derived from soil-to-tissue bioaccumulation models developed for ground insects, not flying insects, which presumably represent the dominant food source for these birds. Overall, potential risks to songbird populations under baseline conditions are considered improbable, but cannot be completely ruled out. Monitoring of COPC concentrations in flying insects as part of the environmental health monitoring program proposed for the mine (Terrestrial Ecosystem Management Plan, 2005) would address this source of uncertainty (see Conclusions and Recommendations in **Section 4**).

Table 6 presents exposure buffers calculated (i.e., 1/HQ) for each ROC/COPC combination. Pending preliminary estimates of increases in COPC concentrations during mine operation (**Section 3.2**), the exposure buffers will provide a basis for commenting on the likelihood of potential increases in COPC-related risks (**Section 3.3**).

3.1.6. Uncertainty Assessment

This section outlines key uncertainties and assumptions made in assessing potential risks to wildlife under baseline (i.e., pre-development) conditions. Specifically, each uncertainty/assumption identified in the exposure and effects assessment is described and evaluated. In addition, potential implications for risk predictions are discussed (e.g., potential for under- or over-estimating risks).

Exposure Assessment

- The modeled ROCs were assumed to represent species that commonly use the Meadowbank area and that will likely be exposed to COPCs once the mine is operational.
- Ground insects were selected to represent tissue concentrations for all invertebrates. These insects were assumed to be representative of the diet of insectivorous birds and mammals during sensitive early development life stages (e.g., feeding of fledglings/juveniles). This approach was assumed to overestimate COPC exposure to birds and mammals that rely primarily on flying insects for their diet.
- COPC uptake efficiencies were assumed to reflect complete contaminant absorption across the gastro-intestinal tract, which is conservative.
- The calculated fractions of methyl mercury were assumed to provide conservative yet realistic, exposure estimates in soil and plant tissues.
- Food consumption, drinking water ingestion, and incidental ingestion of soil were assumed to represent 100% of the total COPC dose for a given receptor. The soil inhalation exposure pathway was assumed to comprise a negligible portion of COPC doses to the ROCs.

Effects Assessment

- Exposure parameters and species-specific TRVs were constructed using available data and best professional judgment to obtain conservative exposure and effects scenarios. It was assumed that the TRVs would be sufficiently conservative to account for uncertainty in the extrapolations being made (e.g., use of subchronic studies; use of mammalian rather than avian toxicity studies; use of studies that measured effects relevant to populations [e.g., reproductive effects] rather than individuals [e.g., behavioural effects]; use of allometric scaling). Ingestion rates were often approximated from similar, but different test species, and parameters were often derived from laboratory experiments instead of field studies. Dietary preferences were often derived from studies on the same or similar species but in a habitat type different from the Meadowbank area.

- It was assumed that the TRVs are protective of populations (LOAEL-based). Avian and mammalian ecotoxicology is best understood for domestic species (e.g., chickens and quail) and/or laboratory test organisms (e.g., mice and rats). The general paucity of data often leads to a conservative approach that relies on the most sensitive study reported in the literature. The ORNL benchmarks upon which most of our TRVs are based have been widely used in risk assessments.
- Potential risks associated with individual COPCs were assessed independently; there was no consideration of potential additive, synergistic or antagonistic effects among COPCs. The uncertainties associated with this assumption are highly dependent on the nature of the multiple contaminant exposures.

In summary, a combination of conservative and realistic decisions was made in the face of uncertainty in the SLRA. Overall, the risk characterization results for wildlife are considered robust.

3.2. Commentary on Likelihood of Increase in COPC Exposure under Future (Post-Development) Conditions

The purpose of this section is to develop preliminary estimates of future concentrations of COPCs in water and soils based on major contamination sources (e.g., dust from tailings piles and other sources, effluent discharge into Third Portage Lake) and discuss potential changes from baseline conditions. This will provide a basis for determining whether baseline exposure buffers will likely be exceeded once the mine is operational (**Section 3.3**).

It is stressed that the data provided in the sections below are preliminary estimates of future concentrations based on models and data provided by other team members. At this time, these preliminary estimates are suggestive that the mine operations are unlikely to result in appreciable increases in metals concentrations in the environment; however, we cannot offer assurance that the predicted environmental concentrations will be representative of future concentrations (i.e., it is possible that concentrations could be either under or overestimated), although very conservative assumptions were generally used. Environmental monitoring of water and soil chemistry is planned as part of environmental management plans for the mine site to verify true environmental concentrations. Appropriate risk management measures will be implemented if concentrations are markedly higher than anticipated or predicted (see Conclusions and Recommendations in **Section 4**).

3.2.1. Water

The evaluation of potential mine-related changes in drinking water quality on the Meadowbank property focused on lake water. Given the abundance and size of the receiving environment lakes as well as their accessibility for all wildlife species, this was considered a reasonable assumption. Other sources of drinking water such as collection ditches, tailings and pit water were not directly assessed. For instance, tailings and pit water will be largely inaccessible to mammals due to steep-sided banks. While these areas will be more accessible to birds, this exposure pathway is likely negligible since birds would need to fly downwards into the pit to access drinking water which is not associated with any sources of food. If necessary, appropriate mitigation measures (e.g., fencing) will be considered to further limit access to tailings water or collection ditches for medium to large mammals (e.g., caribou).

Potential changes in lake water quality might arise from additions of metals as a result of accumulated discharge of effluent and from leaching of metals from rock used to construct the dikes. As indicated in **Section 2.5**, Golder (Water Quality Predictions, 2005) modeled water chemistry changes in Third and Second Portage Lake during each year of mine development, incorporating contributions from effluent and leaching from dike rock using conservative assumptions (e.g., worst-case chemistry, no loss to sediment, accelerated leach rates from lab data, not field cells). **Table 2** presents modeled worst-case predicted water quality results for Third Portage Lake as estimated by Golder. The chemistry of Second Portage Lake, not shown here, is predicted to be the same.

Change to baseline water quality in receiving environment lakes is a function of the baseline concentration, and the loading of a particular contaminant, which is the product of a concentration and rate of discharge. The concentration of nearly all metals in the project lakes are less than laboratory detection limits (BAEAR, 2005) and, with the theoretical exception of cadmium, are well below CCME (2001) guideline concentrations for the protection of aquatic life.

Golder (Water Quality Predictions, 2005) model results indicated that only cadmium was predicted to exceed the CCME (2001) water quality guideline for aquatic life in Third Portage and Second Portage lakes. Only manganese was predicted to exceed the aesthetic objective for drinking water.

The laboratory detection limit for cadmium is 0.00005 mg/L (0.05 parts per billion). Because cadmium in lake water is below this concentration, modeling assumed this concentration for modeling purposes, which is conservative. Thus, maximum model-predicted cadmium concentrations in Third Portage Lake varied from 0.000051 mg/L to 0.000052 mg/L, which exceeds the theoretical CCME guideline for cadmium, based on hardness.

In these project lakes (and other northern, ultra-oligotrophic lake systems), the measured baseline concentrations of cadmium is nearly always less than the detection limit

(<0.00005 mg/L), which is higher than the hardness-derived guideline for cadmium. Obviously, these exceedences are not, in themselves, ecologically relevant. A full discussion of the relevance of the predicted cadmium concentration and its implications is provided in the Aquatic Ecosystem/Fish Habitat Impact Assessment (2005).

Manganese is conservatively predicted to exceed the aesthetic drinking water guideline (i.e., no toxicity is predicted), primarily because of leaching of manganese from dike material placed in the lake. Predicted maximum concentrations of manganese range from 0.009 to 0.072 mg/L, which ranges from below to only 1.4 times higher than the aesthetic objective for this parameter (0.05 mg/L). Simulation results for the two mixing scenarios show that predicted maximum manganese concentrations are below the aesthetic objective when loadings from dikes are excluded for all three scenarios. Results from field cells on site suggested that loading from dikes based on laboratory tests was much less than was used in the model. Also, it was assumed that dike pore water diffused into the lake instead of being pushed through the dike into the pit during the first few years when metal leaching would be greatest.

Overall, water quality in receiving environment lakes was predicted by Golder (Water Quality Predictions, 2005) to remain very good throughout the life of the mine project. Consequently, no significant changes to drinking water quality are expected compared to baseline exposure conditions.

3.2.2. Soil

As described in **Section 2.4**, construction and operation of the Meadowbank Gold Project over its 8 – 10 year life span has the potential to introduce airborne contaminants to soils. As with water, the potential to exceed baseline soil concentrations depends on the mass of particle loss, the chemical composition of the particles, and the spatial extent of dispersal.

Activities with the potential to introduce airborne contaminants will be managed or mitigated to the extent possible. However, there will be unavoidable losses (fugitive emissions) to the environment. Fugitive dust refers to small particles that are of geologic origin that are lost into the atmosphere from point and non-point, uncontrolled sources (Air Quality Impact Assessment, 2005). Dust generation is caused by two main sources, pulverization or abrasion of materials through the application of force (crushing or blasting or rock, truck tires) and from entrainment of particles as a result of wind currents, such as from the waste rock piles or tailings material.

To determine the magnitude (rate of loss) and spatial extent of dispersal of dust from mine site activities, Cumberland (Air Quality Impact Assessment, 2005) conducted a detailed assessment of potential impacts to air quality as one of the supporting documents to the EIA. Information presented here is taken from that document and is used to predict the chemical signature of dust and calculate the magnitude of accumulation in soils surrounding the immediate mine site. Our assessment does not include gaseous emissions, only particulate, as generated from major or important sources. Products of incomplete combustion such as PAHs are not considered here as the Air Quality Impact Assessment (2005) concluded that the volume of these particles produced would be very low and that modeling was not warranted.

According to the Air Quality Impact Assessment (2005), the major contributors to fugitive dust emissions from Meadowbank are:

- **Ore Stockpile** – This stockpile is located near the plant and crusher unit. The pile is predicted to be 17 m high and 43 m in diameter and will be subject to wind erosion. Erosive losses will occur from the stockpile itself and from the conveyor.

- **Tailings Pile** – Wind erosion from the tailings pile is perhaps the most significant source of contaminants. Exposed layers of tailings contain fine particles that will be subject to wind erosion, particularly later in mine life as the pile grows in height and dimension. For modeling purposes, it was assumed that the tailings pile is at least 1 m above the current lake level and fully exposed.

- **Waste Rock Piles** – The Portage and Goose waste rock piles will generate dust from three activities; equipment traffic, unloading and handling and wind erosion of pile surfaces. For modeling purposes, it was assumed that the disposal area was at its maximum size (8.9 ha).

- **Wheel Entrainment** – Dust emissions from large trucks will result from travel along mine site roads and especially back and forth between the Vault deposit and the mill.

- **Drilling and Blasting** – Dust emissions from drilling and blasting were predicted based on standard blast designs. These emissions were not considered significant sources and were not included in the air dispersion model.

All of these activities have the potential to introduce at least small amounts of fugitive dust particles to the atmosphere, where, if conditions are right, they will be carried on the wind and dispersed. Settling depends on particle size, wind speed, barriers (hills, buildings, outcrops) and duration. Metals are the main contaminant carried on such particles.

3.2.2.1. Rate of Spatial Dispersal

Quantitative modeling of dispersal rate (g/m^2/month) from all primary mine sources was conducted at Years 3, 5 and 7 of mine life (Air Quality Impact Assessment, 2005). At different stages of mine life, the size of tailings and waste rock piles will expand and the potential magnitude of loss, increase. Note that local climate conditions (wind speed and direction), temperature and precipitation were incorporated into the modeling exercise. Additionally, it was assumed that there was no dust control of roads to estimate worst-case conditions. Additional details and assumptions about the model are presented in the Air Quality Impact Assessment (2005). Note that only a rate of deposition was calculated, not the chemical composition of dust.

Several scenarios are presented within the Air Quality Impact Assessment (2005). To be conservative we have chosen to represent conditions during Year 7, near the end of mine life as typical for all years (i.e., Years 1 to 8) and use this value to predict accumulation rate over a ten year period, until such time as the tailings pond and waste rock piles are completely capped.

Figure 5 depicts concentration (g/m^2/month) isopleths of particulate deposition from all mine sources during Year 7 of mine life. Concentrations are highest around the ore stockpile, tailings pond and rock storage facility. Isopleths are stretched in a north-south direction, reflecting the prevailing wind direction. Note the exponential decline in concentration of particulate deposition form a maximum of 26 g/m^2/month at "ground zero" at the tailings pile, diminishing to about 2 g/m^2/month at the boundary of the immediate mine site – the Goose Island dike to the south, the waste rock pile to the north and mine site buildings east and west. Concentrations continue to diminish, but at a lower rate with increasing distance from the primary sources.

3.2.2.2. Composition and Accumulation

For the purposes of accumulation, we have assumed a dusting rate of 2 g/m^2/month to the local study area, outside of the immediate mine site area. Furthermore, we have assumed that Year 7 of operation is typical for all years. This is a conservative rate of accumulation as the model predicts lower accumulation rates outside of the mine site. Nevertheless, this approach ensures that accumulation is not underestimated. Therefore, total accumulation in soils is 2 g/month per square meter x 12 months x 10 years or 240 g/m^2 over mine life.

The model presented in the Air Quality Impact Assessment (2005) did not predict chemical composition of the fugitive dust that would accumulate in the environment. Therefore, the likely chemical nature of the dust particles was made according to the following assumptions and is presented in **Table 7**:

- Dust would consist of the three dominant rock types (i.e., IV, UM, and IF).

- Chemistry of these rock types was determined using a four-acid digest. This will liberate more metals than are liberated using a two-acid digest, and is inherently quite conservative.
- The four main sources are from the tailings pile, waste rock piles, ore stockpile and roads. Chemical composition of dust from each source was determined based on the potential source and prorated or weighed according to the relative composition of the rock types. For example, roads are constructed solely from UM rock while waste rock piles will consist of a mixture of rock types.
- Metals are assumed to accumulate in the top two cm of soil, which is completely available to uptake by plants and invertebrates or via incidental ingestion by wildlife. Precipitation (rain and snow melt) and vertical mixing might carry contaminants deeper into the soils; however, given the continuous permafrost, shallow thaw depth (~0.5 – 1.0 m) and absence of worms or other burrowing animals that might mix deeper soils, an assumed 2 cm mixing depth is conservative.

Golder Associates, Burnaby BC determined the chemical composition of IF, UM and IV rocks. Golder (Val Bertrand, Geochemist, Ottawa) was engaged to determine the chemical composition of the four most important sources of dust – the tailings pile, ore stockpile, waste rock pile and roads (see **Section 2.5**; **Table 7**). Golder used a dusting rate of 2 g/m^2/month over a ten-year period to determine the amount (mg) of metals accumulated within the upper 2 cm of soil (adjusted for soil density). Golder assumed roughly equal proportions of dust from each of the four main sources based on the metals concentration of each of the 28 analyzed metals in each potential source.

Metals accumulations in surface (top 2 cm) are predicted to be relatively minor, ranging from less than 1 mg (titanium) to >100 mg for some metals (chromium, manganese, and nickel, reflecting the abundance of these metals in rock). However, relative to the mass of metals in surface soils (baseline soil data; using a 2-acid digest), the relative increase in soil metals is very small, with accumulations of <1% for almost all metals. The only metals to see theoretical increases greater than 1% are barium (2%), nickel (5%), chromium (4%), strontium and vanadium (2%) and thallium (10%; however, baseline thallium was not detectable at the detection limit of <1 mg/kg). These are relatively very small increases in metal loading to local soils.

Potential implications of these predicted changes in exposure from baseline conditions are discussed further in **Section 3.3**.

3.3. Commentary on Potential Risks under Future (Post-Development) Conditions

According to preliminary estimates of future (i.e., post-development) concentrations of COPCs in water and soil, mining activities will likely result in some accumulation of metals in the receiving environment (**Section 3.2**). However, incremental exposure beyond baseline conditions is expected to be very small, specifically:

- No significant changes in drinking water quality are expected compared with current conditions.
- Negligible changes (< 1%) in soil quality are expected for the vast majority of metals. The only metals with theoretical increases greater than 1% are barium (2%), nickel (5%), chromium (4%), strontium and vanadium (2%) and thallium (10%).

Based on these findings, risk predictions made using baseline COPC concentrations are not expected to increase significantly once the mine is operational (see exposure buffers in **Table 6**). As indicated in **Section 3.2**, environmental monitoring of water, soil, and tissue chemistry is planned as part of environmental management plans for the mine site to verify true environmental concentrations (see Conclusions and Recommendations in **Section 4**).

4. CONCLUSIONS AND RECOMMENDATIONS

The approach taken to conduct the wildlife SLRA for the Meadowbank Gold Project required a somewhat unique application of risk assessment. Specifically, the SLRA focused on 1) evaluating whether contaminants pose potential risks to wildlife under baseline exposure conditions, and 2) quantify the magnitude of increase in contaminant exposure required to trigger potential concern for wildlife populations once the mine is operational. Preliminary estimates of future (i.e., post-development) contaminant concentrations were then obtained from models and data provided by other team members. Based on potential future changes in contaminant exposure relative to baseline conditions, implications for potential risks to local wildlife were evaluated.

Key findings and recommendations of the wildlife SLRA are as follows:

- With the exception of chromium, negligible risks were predicted for all COPCs under baseline exposure conditions. For chromium, potential risks to songbird populations (represented by the Lapland longspur) were considered improbable, but could not be completely ruled out. Monitoring of COPC concentrations in flying insects (i.e., the primary medium driving exposure for these birds) as part of the environmental health monitoring program proposed for the mine (Terrestrial Ecosystem Management Plan, 2005) would address this source of uncertainty.
- Since COPC concentrations in water and soil are not expected to change significantly once the mine is operational, potential risks to wildlife are not expected to increase relative to baseline.
- The environmental health monitoring program for the mine should target the following environmental media: soil, water, plants (e.g., sedges, lichens, berries) and flying insects.
- PSVs should be developed to support the environmental health monitoring program. These PSVs would serve to guide decision-making by identifying concentrations below which no unacceptable risks would be anticipated. On the other hand, exceedances of the PSVs would lead to further analysis and/or adaptive management.

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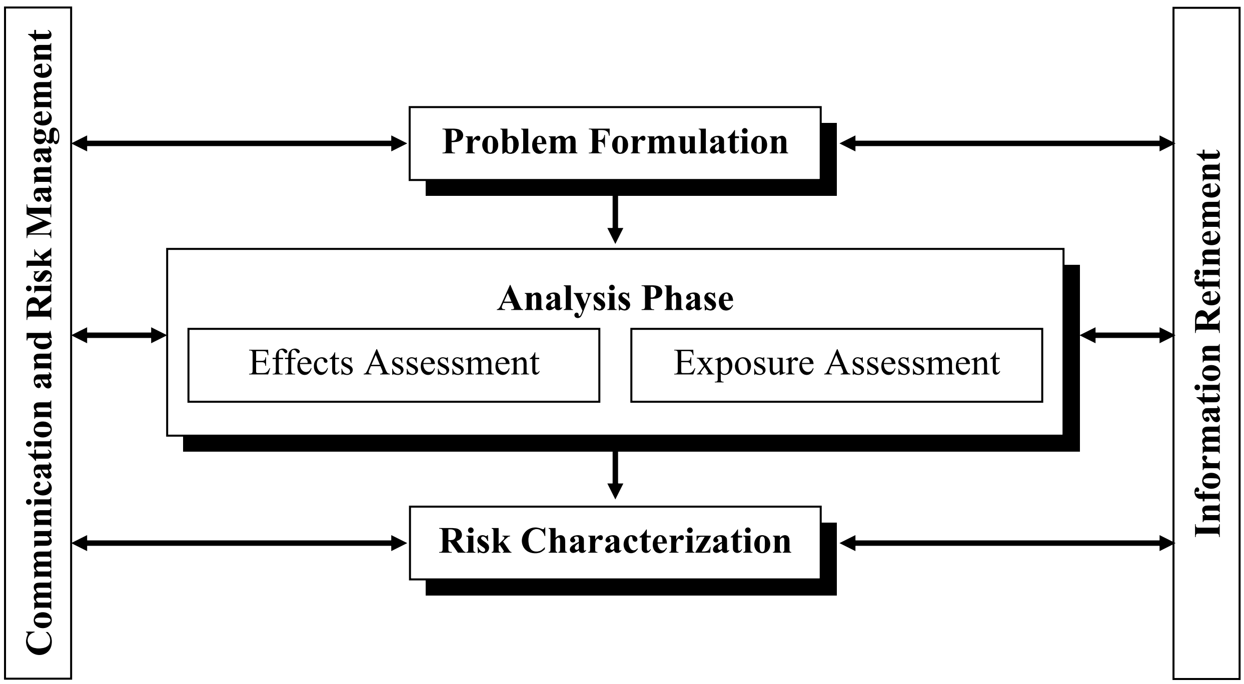
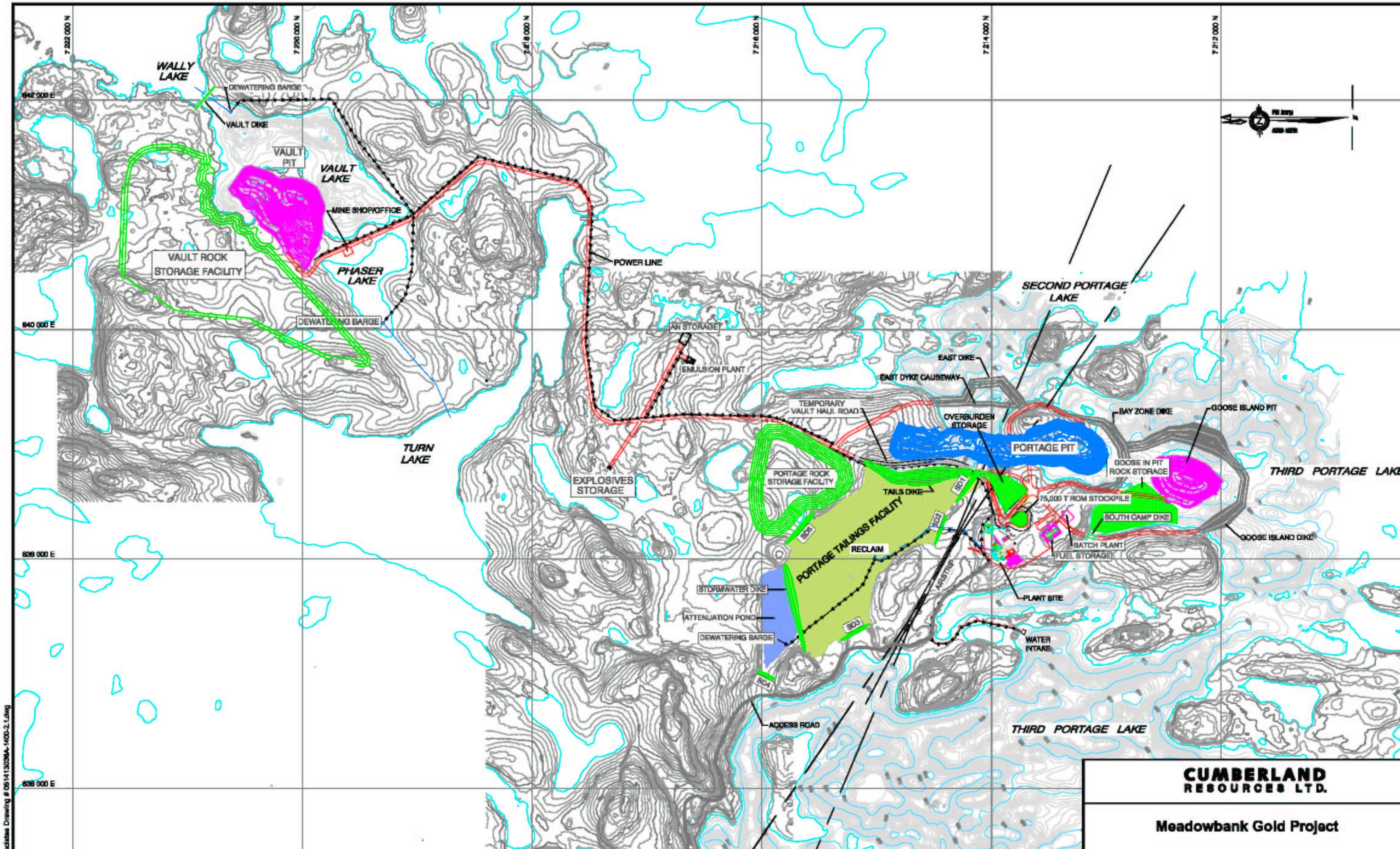
FIGURES

Figure 1. PROJECT LOCATION MAP

Figure 2. ERA Framework





NOTES

1) Topographic contour interval 2m.
2) Bathymetric contour interval 1m.
3) Scale As shown.

LEGEND

SD: SADDLE DAM

CUMBERLAND RESOURCES LTD.

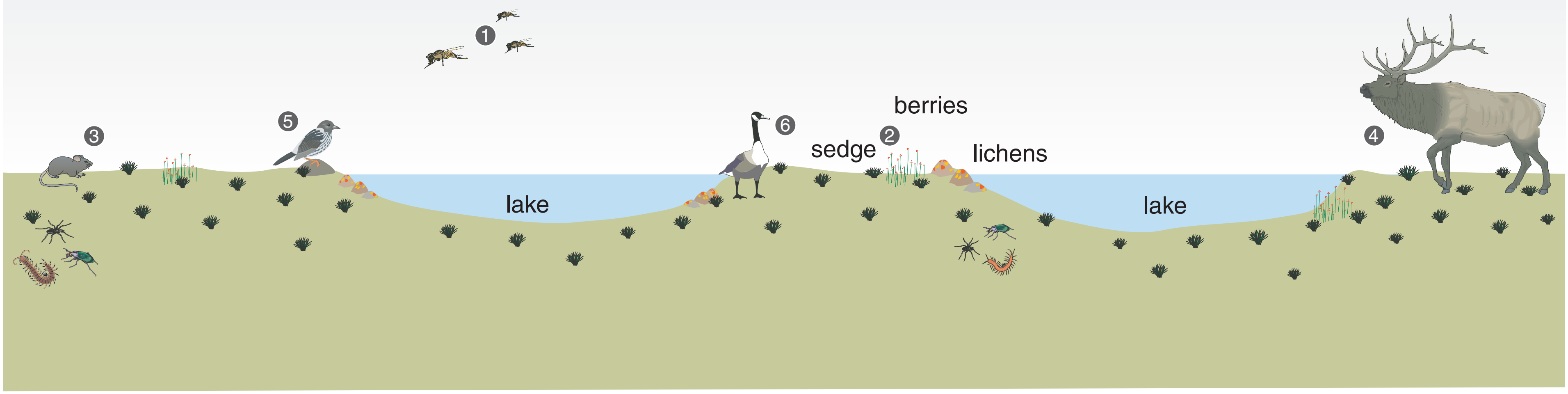
Meadowbank Gold Project

Figure 3. General site plan.

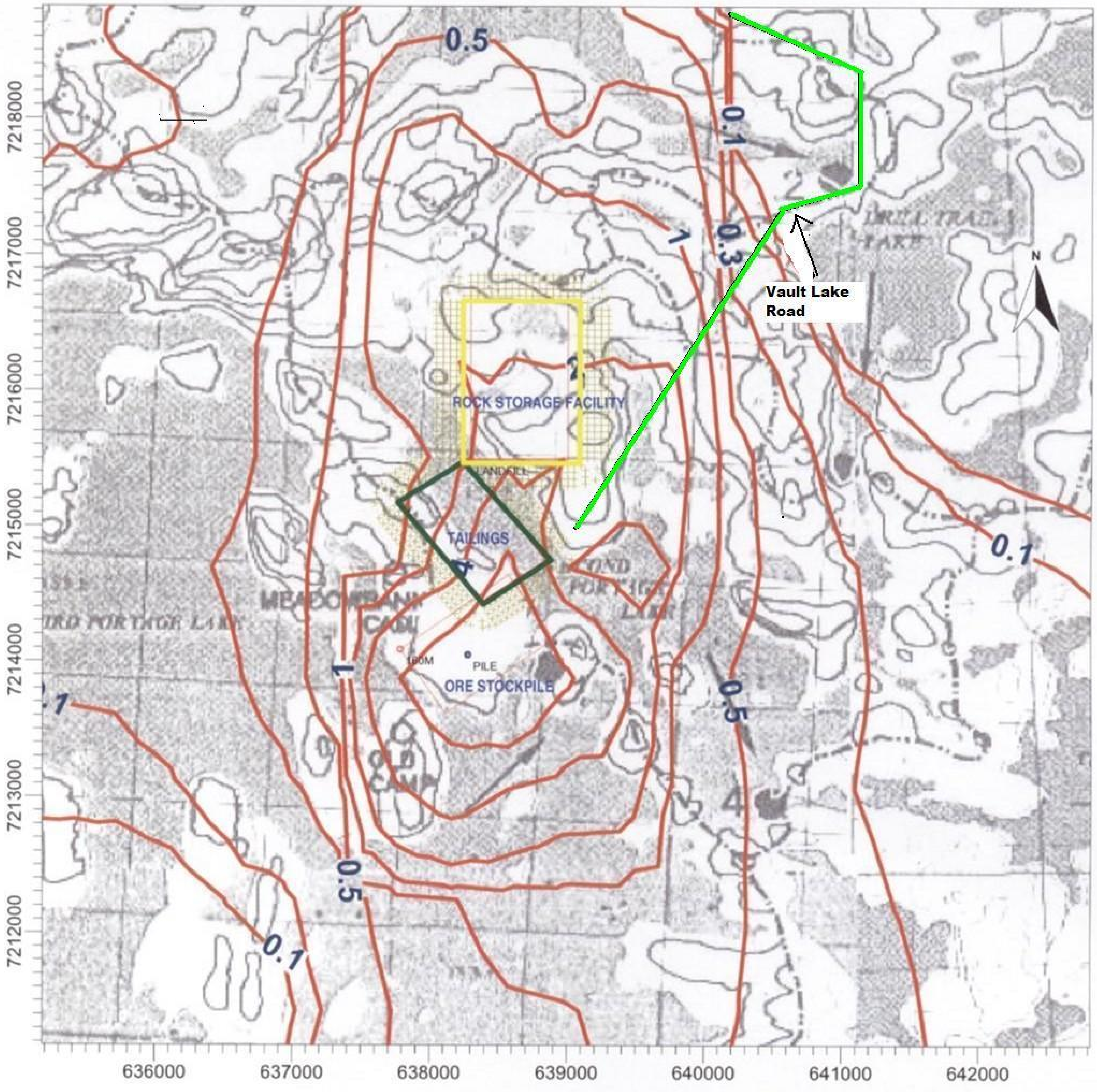
Figure 4. Conceptual model for the Meadowbank Gold Project - Wildlife SLRA

Exposure Pathways

1. Invertebrates (e.g., flying and ground insects) take up COPCs through soil ingestion and direct soil contact.
2. Plants (sedges, lichens, berry producers) uptake COPCs from soil via their root system.
3. Herbivorous small mammals (e.g., northern red-backed vole) may be exposed to COPCs through ingestion of plants, berries, some insects, water, and incidental ingestion of soil.
4. Herbivorous ungulates (e.g., caribou) may be exposed to COPCs through ingestion of plants, water, and incidental ingestion of soil.
5. Omnivorous songbirds (e.g., Lapland longspur) may be exposed to COPCs through ingestion of plants, berries, insects, water, and incidental ingestion of soil.
6. Omnivorous waterfowl (e.g., Canada goose) may be exposed to COPCs through ingestion of plants, berries, insects, water, and incidental ingestion of soil.



Fugitive Dust Dispersion Modelling **(Air Quality Impact Assessment, 2005)**



Maximum dust deposition rate (g/m2/month) from all sources at end of mine life.	MODELING OPTIONS: CONC, DDEP, RURAL, FLAT, TOXICS, DRYDPL	CLIENT NAME: Cumberland Resources Ltd.
	OUTPUT TYPE: DDEP / RECEPTORS: 862	PROJECT NO: EC80864
	MAX: 26.37203 / UNITS: g/m²	DATE: 05/07/2004

0 — 1 km

CUMBERLAND RESOURCES LTD

Figure 5

POST View - Lakes Environmental Software

C:\DISP DUST GOLD\960\MOH\GALL.PLT

TABLES

Table 1. COPC screening table.

Chemical	Units	90th Percentile Baseline Soil Concentrations (mg/kg)	CCME Guidelines	Average Mine Rock and Tailings Chemistry [1] Mine Site Roads	Portage Waste Rock	Vault Waste Rock	Tailings	COPC?
Aluminum	mg/kg	ND	NG	4.0	3.1	7.59	1.16	No
Antimony	mg/kg	10.0	20	**14.87**	6.48	2	5	Yes
Arsenic	mg/kg	24.5	12	1.2	**57.1**	**78**	**115**	Yes
Barium	mg/kg	55.8	500	**79**	**136**	**690**	40	Yes
Beryllium	mg/kg	0.521	4	<0.5	<0.5	**1**	0.5	Yes
Bismuth	mg/kg	ND	NG	<5	-	<5	5	No
Cadmium	mg/kg	0.50	1.4	<1	<1	**2**	<1	Yes
Chromium	mg/kg	124.9	64	**2289.1**	**972.1**	**169**	51	Yes
Cobalt	mg/kg	13.5	40	**73.6**	**32.5**	**16**	10	Yes
Copper	mg/kg	20.0	63	**30.1**	**46.7**	**35**	**182**	Yes
Iron	mg/kg	ND	NG	7.8	8.2	4.73	14.78	No
Gallium	mg/kg	ND	NG	-	-	-	-	No
Lithium	mg/kg	ND	NG	-	-	-	-	No
Lead	mg/kg	30.0	70	20	18	**44**	**46**	Yes
Manganese	mg/kg	436.1	NG	**1244**	**846**	**545**	**475**	Yes
Mercury	mg/kg	0.0825	6.6	0.0087	0.0097	0.005	0	Yes[2]
Molybdenum	mg/kg	4.0	5	<2	<2	**34**	4	Yes
Nickel	mg/kg	60.4	50	**1357**	**470**	**170**	**65**	Yes
Selenium	mg/kg	0.50	1	0.23	-	-	**1**	Yes
Silver	mg/kg	2	20	<1	<1	1	0.4	No
Strontium	mg/kg	25.26	NG	**106**	**79**	**244**	**46**	Yes
Thallium	mg/kg	1	1	-	**34.48**	-	0	Yes
Tin	mg/kg	5	5	-	-	-	<10	No
Titanium	mg/kg	ND	NG	0.17	0.10	0.18	0.02	No
Tungsten	mg/kg	ND	NG	<10	<10	10	<10	No
Uranium	mg/kg	ND	NG	7.3	16.2	-	0	No
Vanadium	mg/kg	25.89	130	**150**	**92**	**99**	**34**	Yes
Zinc	mg/kg	54.8	200	**81**	**75**	**152**	**79**	Yes

[1]Mine Rock chemistry was provided by Golder Associates Ltd. Data represent the average chemistry of each mine component from which dust will be generated during mining activities.

[2] Although mercury concentrations in average mine rock or tailings chemistry did not exceed either baseline soil concentrations or CCME guidelines, mercury was included as a COPC as it is a metal of concern to the general public in the Arctic.

Exceeds CCME Guidelines

Exceeds Baseline Soil Concentrations

Table 2. Predicted Maximum Whole Lake Water Quality, Third Portage Lake (Golder 2005)

Parameter	Expected Loadings (kg/yr)		Average Baseline Conc. (mg/L)[2]	Simulated Maximum Lake Concentration (mg/L)[3]				Water Quality Guideline Concentrations (mg/L)[1] Aquatic Life	NWT Drinking Water Guidelines
				Mid-range Mixing Estimate (92 Mm^3)		Upper Mixing Estimate (169 Mm^3)			
	Water Releases[4]	Dike Leaching				Without Dike Leaching	With Dike Leaching		
Conventional Parameters									
Hardness	21769	8075	5.3	6.0	6.4	5.7	6.0		
pH	n/a	n/a	6.8	n/a	n/a	n/a	n/a	6.5-8.5	
Dissolved Anions									
Total Alkalinity	8924	318	4.0	4.23	4.24	4.13	4.14		
Chloride	14199	589	0.5	1.0	1.1	0.8	0.8	230	250
Fluoride	43	261	0.07	0.07	0.09	0.07	0.08		1.7
Sulphate	18990	501	1.3	2.0	2.0	1.7	1.7		250
Nutrients									
Ammonia Nitrogen	1110	0	0.010	0.0497	0.0497	0.0333	0.0333	7.2	
Total Kjeldahl Nitrogen	n/a	n/a	0.09	n/a	n/a	N/a	n/a	n/a	n/a
Nitrate Nitrogen	1465	42	<0.0040	0.0569	0.0588	0.0351	0.0363	2.9	45
Nitrite Nitrogen	n/a	n/a	0.0010	n/a	n/a	N/a	n/a		
Total Phosphate-P	10.9	0.0	0.0020	0.0024	0.0024	0.0022	0.0022		
Total Phosphorus	34.3	6.2	0.002	0.0032	0.0035	0.0027	0.0029		
Organic Parameters									
Dissolved Organic Carbon	n/a	n/a	1.4	n/a	n/a	N/a	n/a		
Cyanides									
Total Cyanide	0	0	<0.005	0	0	0	0	0.005	
Total Metals									
Aluminum	38	50	0.006	0.007	0.01	0.007	0.009	0.1	
Antimony	3.2	0.28	<0.0005	0.0006	0.00062	0.00056	0.00057		
Arsenic	6.1	0.04	<0.0005	0.00072	0.00072	0.00062	0.00062	0.005	0.05
Barium	21	32.8	<0.02	0.02	0.023	0.02	0.022		1
Beryllium	1.38	0.00	<0.001	0.001	0.001	0.001	0.001		
Boron	178	0.0	<0.1	0.104	0.104	0.102	0.102		
Bismuth	0.25	0.00		0.00001	0.00001	0.00001	0.00001		
Cadmium[1]	0.10	0.003	<0.00005	**<0.000052**	**<0.000052**	**<0.000051**	**<0.000051**	0.00010 – 0.00017 (EPA) 0.0000026 to 0.000017 CCME)	0.01
Calcium	7592	1692	1.2	1.5	1.5	1.3	1.4		

Table 2 – Continued

Parameter	Expected Loadings (kg/yr) Water Releases[4]	Dike Leaching	Average Baseline Conc. (mg/L)[2]	Simulated Maximum Lake Concentration (mg/L)[3] Mid-range Mixing Estimate (92 Mm^3)		Upper Mixing Estimate (169 Mm^3) Without Dike Leaching	With Dike Leaching	Water Quality Guideline Concentrations (mg/L)[1] Aquatic Life	NWT Drinking Water Guidelines
Chromium	0.4	0.006	<0.001	0.001	0.001	0.001	0.001	0.001/0.0089	0.05
Cobalt	3.24	17.84	<0.0003	0.0004	0.0017	0.004	0.0013		
Copper	9.0	0.26	<0.001	0.0013	0.0013	0.0012	0.0012	0.002	1
Iron	44.3	10.0	<0.03	0.03	0.03	0.03	0.03	0.3	0.3
Lead	1.8	0.1	0.0006	0.0006	0.0007	0.0006	0.0006	0.001	0.05
Lithium	2.3	0.0	<0.0050	0.005	0.005	0.005	0.005		
Magnesium	4056	935	0.5	0.6	0.7	0.6	0.6		
Manganese	386	841	0.001	0.015	**0.072**	0.009	**0.052**		0.05
Mercury	0.029	0.056	<0.0001	0.00005	0.00005	0.00005	0.00005	0.0001	
Molybdenum	2.8	0.06	<0.001	0.001	0.001	0.001	0.001	0.073	
Nickel	27.6	1.6	<0.001	0.002	0.0021	0.0016	0.0016	0.025	
Potassium	1826	892	<2	2.0	2.1	2.0	2.1		
Selenium	0.62	0.01	<0.001	0.001	0.001	0.001	0.001	0.001	0.01
Silver	0.014	0.039	<0.00002	0.00002	0.00002	0.00002	0.00002	0.0001	0.05
Silicon	568	1380		0.02	0.12	0.01	0.08		
Sodium	2578	563	2.0	2.0	2.1	2.0	2.0		
Strontium	108	70		0.004	0.007	0.002	0.005		
Thallium	0.07	0.00	<0.0002	0.0002	0.0002	0.0002	0.0002	0.0008	
Tin	n/a	n/a	<0.0006	n/a	n/a	n/a	n/a		
Titanium	n/a	n/a	<0.01	n/a	n/a	n/a	n/a		5
Uranium	0.77	0.7	<0.0002	0.0002	0.0003	0.0002	0.0002		0.02
Vanadium	4.0	1.1	<0.03	0.03	0.03	0.03	0.03		
Zinc	272	1.3	0.005	0.015	0.015	0.011	0.011	0.03	5

Notes: 1. Canadian Water Quality Guidelines are used, except for aquatic life.. Both CWQG and US EPA values are shown for cadmium, with range of published value to hardness adjusted. **2.** Total metals. **3.** Dissolved metals. **4.** Average loadings per year based on predicted data for the 8 years of operations. Maximum predicted arsenic concentrations are predicted to vary from 0.003 to 0.029 mg/L, which ranges from less than the guideline to six times the guideline of 0.005 mg/L. Maximum predicted nickel concentrations are predicted to vary from 0.005 to 0.035 mg/L, which ranges from less than the guideline to 1.6 times the guideline of 0.025 mg/L. [5] Simulated maximum lake concentrations based on worst case predictions and are expected to be higher than those based on expected case predictions. Values in **bold** indicate exceedance of the WQGC and NWT drinking water guidelines.
Source: Golder (2005b)

Table 3. COPC summary statistics for baseline soil, lake water and tissue data.

	Units	pH	Moisture Content (%)	Antimony	Arsenic	Barium	Beryllium	Cadmium	Chromium	Cobalt	Copper	Lead
Sedges												
Sample Size	count	-	50	50	50	50	50	50	50	50	50	50
Minimum	mg/kg dw	-	28.20	0.05	0.05	18.90	0.30	0.03	0.50	0.13	1.41	0.22
Maximum	mg/kg dw	-	79.90	0.26	10.30	93.80	0.45	0.98	87.10	6.10	25.10	14.50
Mean	mg/kg dw	-	51.26	0.05	0.67	39.28	0.30	0.17	7.59	1.21	5.61	1.55
Standard Deviation	mg/kg dw	-	13.71	0.03	1.72	17.71	0.02	0.22	15.30	1.61	4.06	2.69
90th Percentile	mg/kg dw	-	69.87	0.05	1.60	66.47	0.30	0.34	16.09	3.67	9.51	3.41
95% UCLM	mg/kg dw	-	54.44	0.06	1.09	43.24	0.31	0.22	11.09	1.59	6.55	2.15
Lichen												
Sample Size	count	-	50	50	50	50	50	50	50	50	50	50
Minimum	mg/kg dw	-	1.20	0.05	0.08	11.60	0.30	0.07	0.66	0.11	1.32	0.52
Maximum	mg/kg dw	-	29.80	0.05	1.48	48.20	0.30	0.33	9.97	1.94	10.30	5.14
Mean	mg/kg dw	-	11.77	0.05	0.32	24.52	0.30	0.15	2.97	0.48	2.44	2.53
Standard Deviation	mg/kg dw	-	6.57	0.00	0.29	8.22	0.00	0.05	2.12	0.33	1.72	1.10
90th Percentile	mg/kg dw	-	21.29	0.05	0.60	34.07	0.30	0.21	5.53	0.87	2.94	3.98
95% UCLM	mg/kg dw	-	13.22	0.05	0.39	26.46	0.30	0.17	3.48	0.56	2.85	2.78
Berries												
Sample Size	count	-	50	50	50	50	50	50	50	50	50	50
Minimum	mg/kg dw	-	71.90	0.05	0.05	4.45	0.30	0.03	0.50	0.10	4.01	0.10
Maximum	mg/kg dw	-	98.40	0.05	0.05	19.70	0.30	0.30	0.50	0.11	7.91	0.21
Mean	mg/kg dw	-	86.15	0.05	0.05	10.85	0.30	0.12	0.50	0.10	5.24	0.10
Standard Deviation	mg/kg dw	-	3.17	0.00	0.00	3.79	0.00	0.07	0.00	0.00	0.72	0.02
90th Percentile	mg/kg dw	-	88.71	0.05	0.05	15.53	0.30	0.22	0.50	0.10	6.01	0.10
95% UCLM	mg/kg dw	-	86.83	0.05	0.05	11.71	0.30	0.14	0.50	0.10	5.41	0.11

Table 3. COPC summary statistics for baseline soil, lake water and tissue data.

	Units	pH	Moisture Content (%)	Antimony	Arsenic	Barium	Beryllium	Cadmium	Chromium	Cobalt	Copper	Lead
Soil												
Sample Size	count	50	-	50	50	50	50	50	50	50	50	50
Minimum	mg/kg dw	4.39	-	10	5.0	20.3	0.50	0.50	8.6	2.3	2.9	30.0
Maximum	mg/kg dw	6.79	-	10	173.0	174.0	1.13	0.62	193.0	16.4	26.1	55.0
Mean	mg/kg dw	5.58	-	10	12.6	43.9	0.52	0.51	58.0	7.8	10.2	30.5
Standard Deviation	mg/kg dw	0.65	-	0	24.9	25.3	0.10	0.02	47.4	3.8	6.2	3.5
90th Percentile	mg/kg dw	6.36	-	10	24.5	55.8	0.52	0.50	124.9	13.5	20.0	30.0
95% UCLM	mg/kg dw	5.73	-	10	18.4	49.8	0.55	0.51	69.2	8.7	11.6	31.4
Lake Water												
Sample Size	count	-	-	51	51	51	51	51	51	51	51	51
Minimum	mg/L	-	-	0.0001	0.0001	0.002	0.001	0.0001	0.0005	0.0001	0.0003	0.0001
Maximum	mg/L	-	-	0.0005	0.0005	0.020	0.005	0.0058	0.0010	0.0010	0.0010	0.0039
Mean	mg/L	-	-	0.0004	0.0004	0.015	0.001	0.0002	0.0009	0.0003	0.0009	0.0006
Standard Deviation	mg/L	-	-	0.0002	0.0002	0.008	0.001	0.0008	0.0002	0.0002	0.0002	0.0007
90th Percentile	mg/L	-	-	0.0005	0.0005	0.020	0.001	0.0002	0.0010	0.0003	0.0010	0.0009
95% UCLM	mg/L	-	-	0.0004	0.0004	0.015	0.001	0.0003	0.0008	0.0003	0.0008	0.0006

Table 3. COPC summary statistics for baseline soil, lake water and tissue data.

	Units	Manganese	Mercury	Molybdenum	Nickel	Selenium	Strontium	Thallium	Vanadium	Zinc
Sedges										
Sample Size	count	50	50	50	50	50	50	50	50	50
Minimum	mg/kg dw	147.00	0.02	0.20	0.95	1.00	5.27	0.03	0.50	15.20
Maximum	mg/kg dw	1040.00	0.08	8.28	62.90	2.00	29.40	0.10	13.60	85.60
Mean	mg/kg dw	390.18	0.04	1.43	9.35	1.02	14.08	0.03	1.51	43.52
Standard Deviation	mg/kg dw	181.07	0.02	1.54	12.96	0.14	6.09	0.01	2.54	15.24
90th Percentile	mg/kg dw	641.80	0.06	2.26	17.79	1.00	24.06	0.04	3.00	60.78
95% UCLM	mg/kg dw	431.53	0.04	1.79	12.22	1.05	15.46	0.04	2.12	46.94
Lichen										
Sample Size	count	50	50	50	50	50	50	50	50	50
Minimum	mg/kg dw	62.90	0.03	0.05	0.65	1.00	4.46	0.03	0.50	14.10
Maximum	mg/kg dw	298.00	0.25	0.37	40.30	1.00	21.40	0.04	2.59	37.60
Mean	mg/kg dw	171.80	0.10	0.12	3.16	1.00	9.61	0.03	0.85	24.07
Standard Deviation	mg/kg dw	65.50	0.04	0.06	5.54	0.00	3.72	0.00	0.49	4.85
90th Percentile	mg/kg dw	262.90	0.14	0.19	4.56	1.00	14.09	0.03	1.45	29.76
95% UCLM	mg/kg dw	187.58	0.11	0.14	4.42	1.00	10.46	0.03	0.96	25.22
Berries										
Sample Size	count	50	50	50	50	50	50	50	50	50
Minimum	mg/kg dw	32.10	0.01	0.05	0.80	1.00	1.17	0.03	0.50	7.40
Maximum	mg/kg dw	357.00	0.07	0.50	3.82	1.00	5.69	0.03	0.50	31.60
Mean	mg/kg dw	117.45	0.01	0.14	1.85	1.00	2.70	0.03	0.50	15.17
Standard Deviation	mg/kg dw	70.11	0.01	0.10	0.76	0.00	0.97	0.00	0.00	5.56
90th Percentile	mg/kg dw	193.30	0.01	0.26	2.82	1.00	4.03	0.03	0.50	21.78
95% UCLM	mg/kg dw	133.96	0.01	0.16	2.03	1.00	2.92	0.03	0.50	16.50

Table 3. COPC summary statistics for baseline soil, lake water and tissue data.

	Units	Manganese	Mercury	Molybdenum	Nickel	Selenium	Strontium	Thallium	Vanadium	Zinc
Soil										
Sample Size	count	50	50	50	50	50	50	50	50	50
Minimum	mg/kg dw	135	0.0050	4.0	5.0	0.10	7.63	1.0	4.8	20.8
Maximum	mg/kg dw	721	0.3560	4.0	97.3	0.50	39.80	1.0	32.5	64.5
Mean	mg/kg dw	320	0.0393	4.0	29.3	0.49	15.46	1.0	15.9	38.6
Standard Deviation	mg/kg dw	106	0.0579	0.0	23.0	0.06	7.49	0.0	6.8	11.0
90th Percentile	mg/kg dw	436	0.0825	4.0	60.4	0.50	25.26	1.0	25.9	54.8
95% UCLM	mg/kg dw	345	0.0534	4.0	34.5	0.51	17.24	1.0	17.5	41.0
Lake Water										
Sample Size	count	51	51	51	51	51	15	51	51	51
Minimum	mg/L	0.0006	0.0001	0.0001	0.0002	0.001	0.0045	0.0001	0.001	0.001
Maximum	mg/L	0.0050	0.0001	0.0010	0.0010	0.001	0.0120	0.1000	0.030	0.023
Mean	mg/L	0.0015	0.0001	0.0008	0.0008	0.001	0.0072	0.0041	0.023	0.005
Standard Deviation	mg/L	0.0010	0.0000	0.0004	0.0003	0.000	0.0026	0.0196	0.013	0.004
90th Percentile	mg/L	0.0023	0.0001	0.0010	0.0010	0.001	0.0109	0.0002	0.030	0.006
95% UCLM	mg/L	0.0016	0.0000	0.0008	0.0008	0.001	0.0026	0.0078	0.023	0.006

Table 4. Terrestrial toxicity reference values (TRVs) for birds and mammals.

Parameter				Antimony[2,3,4]	Arsenic[1]	Barium[1]	Beryllium[1,2]	Cadmium[1]	Chromium[1,5]	Cobalt[7]
TRVs for Mammals (see allometric scaling equation in footnotes)										
	NOAEL-based TRV:	Test Species		Mouse	Mouse	Rat	Rat	Rat	Rat	Rat
		BW_{NOAEL}(kg wet)		0.03	0.03	0.435	0.35	0.303	0.35	0.15
		NOAEL (mg/kg wet/day)		98.0	0.126	5.1	0.66	1	3.28	0.2
	LOAEL-based TRV:	Test Species		Rat	Mouse	Rat	na	Rat	Rat	Rabbit
		BW_{LOAEL}(kg wet)		0.27	0.03	0.35	na	0.303	0.35	3
		LOAEL (mg/kg wet/day)		112.9	1.26	19.8	na	10	13.14	2
TRVs for Birds (allometric scaling factor of 1 was assumed; see footnotes)										
	NOAEL-based TRV:	Test Species		Rat (see above)	Brown-headed cowbird	Chicken	Rat (see above)	Mallard	Black duck	Pek. Duckling
		NOAEL (mg/kg wet/day)		9.8	2.5	21	0.066	1.5	1	2.37
	LOAEL-based TRV:	Test Species		Rat (see above)	Brown-headed cowbird	Chicken	na	Mallard	Black duck	Pek. Duckling
		LOAEL (mg/kg wet/day)		11.29	7.4	42	na	20	5	4.74
	Wildlife Species	**Body Weight (kg wet)**								
Mammals	Northern Red-backed Vole	0.02	NOAEL	108.45	0.14	11.01	1.35	1.97	6.71	0.33
	Northern Red-backed Vole	0.02	LOAEL	216.41	1.39	40.50	na	19.73	26.88	7.00
	Caribou	75	NOAEL	13.86	0.02	1.41	0.17	0.25	0.86	0.04
	Caribou	75	LOAEL	27.65	0.18	5.18	na	2.52	3.43	0.89
Birds	Lapland Longspur	0.023	NOAEL	9.80	2.46	21	0.066	1.45	1.00	2.37
	Lapland Longspur	0.023	LOAEL	11.29	7.38	42	na	20.03	5.00	4.74
	Canada Goose	2	NOAEL	9.80	2.46	21	0.066	1.45	1.00	2.37
	Canada Goose	2	LOAEL	11.29	7.38	42	na	20.03	5.00	4.74

Notes:
Based on Sample et al. (1996), the following allometric equation was used for interspecies extrapolations among mammals:
$NOAEL_w = NOAEL_{ts} * (BW_{ts}/BW_w)$^0.25; the equation also applies to the LOAEL
Based on Sample et al. (1996), an allometric scaling factor of 1 was considered appropriate for interspecies extrapolations among birds
underline corresponds to an unbounded LOAEL (10X safety factor used to derive the NOAEL) (see text for details)
na indicates that there was no TRV (NOAEL or LOAEL) available

[1] Sample et al. (1996)
[2] Bird TRVs calculated by multiplying the mammal TRVs with a safety factor of 0.1 (see text for discussion)
[3] NOAEL from Dieter et al. (1991) as quoted in Lynch et al. (1999)
[4] LOAEL from Rossi et al. (1987)
[5] Mammals TRV based on chromium VI; bird TRV based on chromium III
[6] Ueberschar et al. (1986)
[7]Chetty et al. (1979) for mammal NOAEL TRV, Szakmary et al. (2001) for mammal LOAEL TRV, Van Vleet (1982) for bird TRVs.

Table 4. Terrestrial toxicity reference values (TRVs) for birds and mammals.

Parameter				Copper[1]	Lead[1]	Manganese[1]	Total Hg	Inorg-Hg[1]	MeHg[1]	Molybdenum[1]	Nickel[1]
TRVs for Mammals (see allometric scaling equation in footnotes)											
	NOAEL-based TRV:	Test Species		Mink	Rat	Rat	na	Mink	Mink	Mouse	Rat
		BW_{NOAEL}(kg wet)		1	0.35	0.35	na	1	1	0.03	0.35
		NOAEL (mg/kg wet/day)		11.7	8	88	na	1	0.015	<u>0.26</u>	40
	LOAEL-based TRV:	Test Species		Mink	Rat	Rat	na	Mink	Mink	Mouse	Rat
		BW_{LOAEL}(kg wet)		1	0.35	0.35	na	1	1	0.03	0.35
		LOAEL (mg/kg wet/day)		15.14	80	284	na	na	0.025	2.6	80
TRVs for Birds (allometric scaling factor of 1 was assumed; see footnotes)											
	NOAEL-based TRV:	Test Species		Chicken	Japanese quail	Japanese quail	na	Japanese quail	Mallard	Chicken	Mallard
		NOAEL (mg/kg wet/day)		47	1.13	977	na	0.45	0.0064	<u>3.53</u>	77.4
	LOAEL-based TRV:	Test Species		Chicken	Japanese quail	na	na	Japanese quail	Mallard	Chicken	Mallard
		LOAEL (mg/kg wet/day)		62	11.3	na	na	0.9	0.064	35.3	107
	Wildlife Species	**Body Weight (kg wet)**									
Mammals	Northern Red-backed Vole	0.02	NOAEL	31.11	16.36	179.99	na	2.66	0.04	0.29	81.81
	Northern Red-backed Vole	0.02	LOAEL	40.26	163.62	580.87	na	na	0.07	2.88	163.62
	Caribou	75	NOAEL	3.98	2.09	23.00	na	0.34	0.01	0.04	10.45
	Caribou	75	LOAEL	5.14	20.91	74.23	na	na	0.01	0.37	20.91
Birds	Lapland Longspur	0.023	NOAEL	46.97	1.13	977.00	na	0.45	0.01	3.53	77.40
	Lapland Longspur	0.023	LOAEL	61.72	11.30	na	na	0.90	0.06	35.30	107.00
	Canada Goose	2	NOAEL	46.97	1.13	977.00	na	0.45	0.01	3.53	77.40
	Canada Goose	2	LOAEL	61.72	11.30	na	na	0.90	0.06	35.30	107.00

Notes:
Based on Sample
et al. (1996), the
Based on Sample et al. (1996), an allometric scaling factor of 1 was considered appropriate for interspecies extrapolations among birds
<u>underline</u> corresponds to an unbounded LOAEL (10X safety factor used to derive the NOAEL) (see text for details)
na indicates that there was no TRV (NOAEL or LOAEL) available

[1] Sample et al. (1996)
[2] Bird TRVs calculated by multiplying the mammal TRVs with a safety factor of 0.1 (see text for discussion)
[3] NOAEL from Dieter et al. (1991) as quoted in Lynch et al. (1999)
[4] LOAEL from Rossi et al. (1987)
[5] Mammals TRV based on chromium VI; bird TRV based on chromium III
[6] Ueberschar et al. (1986)
[7]Chetty et al. (1979) for mammal NOAEL TRV, Szakmary et al. (2001) for mammal LOAEL TRV, Van Vleet (1982) for bird TRVs.

Table 4. Terrestrial toxicity reference values (TRVs) for birds and mammals.

Parameter				Selenium[1]	Strontium[1,2]	Thallium[1,6]	Vanadium[1]	Zinc[1]
TRVs for Mammals (see allometric scaling equation in footnotes)								
	NOAEL-based TRV:	Test Species		Rat	Rat	Rat	Rat	Rat
		BW_{NOAEL}(kg wet)		0.35	0.35	0.365	0.26	0.35
		NOAEL (mg/kg wet/day)		0.2	263	<u>0.0074</u>	<u>0.21</u>	160
	LOAEL-based TRV:	Test Species		Rat	na	Rat	Rat	Rat
		BW_{LOAEL}(kg wet)		0.35	na	0.365	0.26	0.35
		LOAEL (mg/kg wet/day)		0.33	na	0.074	2.1	320
TRVs for Birds (allometric scaling factor of 1 was assumed; see footnotes)								
	NOAEL-based TRV:	Test Species		Mallard	Rat (see above)	Chicken	Mallard	White leghorn hen
		NOAEL (mg/kg wet/day)		0.4	26.3	0.202	11.4	14.5
	LOAEL-based TRV:	Test Species		Mallard	na	Chicken	Mallard	White leghorn hen
		LOAEL (mg/kg wet/day)		0.8	na	0.757	na	131
	Wildlife Species	**Body Weight (kg wet)**						
Mammals	Northern Red-backed Vole	0.02	NOAEL	0.41	537.92	0.02	0.40	327.25
	Northern Red-backed Vole	0.02	LOAEL	0.67	na	0.15	3.99	654.50
	Caribou	75	NOAEL	0.05	68.74	0.002	0.05	41.82
	Caribou	75	LOAEL	0.09	na	0.02	0.51	83.64
Birds	Lapland Longspur	0.023	NOAEL	0.40	26.3	na	na	14.49
	Lapland Longspur	0.023	LOAEL	0.80	na	na	na	130.90
	Canada Goose	2	NOAEL	0.40	26.3	na	na	14.49
	Canada Goose	2	LOAEL	0.80	na	na	na	130.90

Notes:
Based on Sample
et al. (1996), the
Based on Sample et al. (1996), an allometric scaling factor of 1 was considered appropriate for interspecies extrapolations among birds
<u>underline</u> corresponds to an unbounded LOAEL (10X safety factor used to derive the NOAEL) (see text for details)
na indicates that there was no TRV (NOAEL or LOAEL) available

1 Sample et al. (1996)
2 Bird TRVs calculated by multiplying the mammal TRVs with a safety factor of 0.1 (see text for discussion)
3 NOAEL from Dieter et al. (1991) as quoted in Lynch et al. (1999)
4 LOAEL from Rossi et al. (1987)
5 Mammals TRV based on chromium VI; bird TRV based on chromium III
6 Ueberschar et al. (1986)
7 Chetty et al. (1979) for mammal NOAEL TRV, Szakmary et al. (2001) for mammal LOAEL TRV, Van Vleet (1982) for bird TRVs.

Table 5. Summary of hazard quotients (HQs) for each COPC/ROC combination.

	ROCs			
COPCs	**Northern red-backed vole**	**Caribou**	**Lapland longspur**	**Canada goose**
Antimony	0.0002	0.0002	0.149	0.0013
Arsenic	0.05	0.1	0.05	0.003
Barium	0.05	0.2	0.28	0.01
Beryllium	0.01	0.05	1.8	0.06
Cadmium	0.0006	0.002	0.009	0.0002
Chromium	0.02	0.05	2.5	0.04
Cobalt	0.013	0.03	0.34	0.005
Copper	0.008	0.02	0.05	0.001
Lead	0.0007	0.004	0.06	0.004
Manganese	0.03	0.09	0.10	0.004
Total Hg	na	na	na	na
Inorg-Hg	0.0005	0.005	0.012	0.0003
MeHg	0.008	0.10	0.02	0.002
Molybdenum	0.03	0.04	0.023	0.0006
Nickel	0.004	0.01	0.06	0.001
Selenium	0.07	0.3	0.2	0.01
Strontium	0.0012	0.005	0.16	0.006
Thallium	0.05	0.10	na	na
Vanadium	0.04	0.08	na	na
Zinc	0.004	0.01	0.3	0.004

Notes:
na - not available

underline indicates that the HQ was calculated using the NOAEL-based TRV rather than the LOAEL-based TRV (i.e., when LOAEL-based TRVs were not available)

Table 6. Magnitude of increase in COPC exposure required to trigger potential concerns for wildlife populations.

	ROCs			
COPCs	**Northern red-backed vole**	**Caribou**	**Lapland longspur**	**Canada goose**
Antimony	5796	4058	7	769
Arsenic	22	7	19	296
Barium	22	6	4	97
Beryllium	87	20	1	17
Cadmium	1638	496	112	6663
Chromium	42	20	0	27
Cobalt	79	36	3	190
Copper	124	50	21	773
Lead	1381	238	16	252
Manganese	33	11	10	237
Total Hg	na	na	na	na
Inorg-Hg	2131	203	80	2941
MeHg	121	10	50	533
Molybdenum	36	25	43	1767
Nickel	283	117	16	723
Selenium	14	3	5	73
Strontium	838	217	6	174
Thallium	22	10	na	na
Vanadium	27	12	na	na
Zinc	285	102	4	245

Notes:
na - not available

underline indicates that the exposure buffer was calculated using the NOAEL-based HQ rather than the LOAEL-based HQ (i.e., when LOAEL-based TRVs were not available)

Table 7. Estimated relative Increase in metals concentrations in soil due to dust generation during mine operations

Chemical	Units	90th Percentile Baseline Soil Concentrations (mg/kg)	Mean Metals in Mine Rock Dust Sources[1]	mass (mg) of constituent in 10-year accumulation of dust	mass (mg) of constituent in top 2cm of soil	constituent concentration in top 2cm soil after 10-yrs mining	Relative increase (%)
Antimony	mg/kg	10.0	7.1	1.70	390	10.0	0%
Arsenic	mg/kg	24.5	62.8	15.08	955	24.7	1%
Barium	mg/kg	55.8	236.4	56.75	2176	56.9	2%
Beryllium	mg/kg	0.521	*0.6*	0.15	20	0.5	0%
Cadmium	mg/kg	0.50	*1.3*	0.30	20	0.5	1%
Chromium	mg/kg	124.9	870.3	208.88	4871	129.5	4%
Cobalt	mg/kg	13.5	33.0	7.92	525	13.6	1%
Copper	mg/kg	20.0	73.5	17.63	778	20.3	2%
Lead	mg/kg	30.0	32.1	7.71	1170	30.0	0%
Manganese	mg/kg	436.1	777.4	186.57	17008	438.2	0%
Mercury	mg/kg	0.0825	0.008	0.00	3	0.1	-1%
Molybdenum	mg/kg	4.0	*10.5*	2.52	156	4.0	1%
Nickel	mg/kg	60.4	515.5	123.71	2354	63.2	5%
Selenium	mg/kg	0.50	0.6	0.15	20	0.5	0%
Strontium	mg/kg	25.26	118.8	28.50	985	25.8	2%
Thallium	mg/kg	1	17.2	4.14	39	1.1	10%
Vanadium	mg/kg	25.89	93.8	22.51	1010	26.3	2%
Zinc	mg/kg	54.8	96.9	23.25	2136	55.0	0%

Notes:

equals method detection limit

[1]Mine Rock chemistry was provided by Golder Associates Ltd. Data represent the average chemistry of four sources of mine rock (portage/goose waste rock, vault waste rock, tailings and road rock) from which dust is likely to be generated during mine operations.The chemistry of dust particles is assumed to be the same as that of the parent rock material.

Dusting Rate	240	g/m2/10-year mine life
Dry density of soil	1950	kg/m3
assumed moisture content	20	%
wet density of soil	2150	kg/m3
Volume of soil in 2cm layer/m2	0.02	m3
mass of dry soil in top 2cm	39	kg

n/c : not calculated