Exponent®

DMTS Fugitive Dust Risk Assessment Work Plan

Prepared for

Teck Cominco Alaska Incorporated
Anchorage, Alaska

Exponent®

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Risk Assessment Work Plan

Prepared for

Teck Cominco Alaska Incorporated
3105 Lakeshore Drive
Building A, Suite 101
Anchorage, AK 99517

Prepared by

Exponent
15375 SE 30th Place, Suite 250
Bellevue, WA 98007

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Contents

List of Figures

Figures are presented at the end of the main text.

List of Tables

Tables are presented at the end of the main text.

List of Photographs

Photographs are presented at the end of the main text.

Acronyms and Abbreviations

AIDEA	Alaska Industrial Development and Export Authority
ANOVA	analysis of variance
AWQC	ambient water quality criteria
CCC	criterion continuous concentration
CDC	Centers for Disease Control and Prevention
CMC	criteria maximum concentration
CoPC	chemical of potential concern
CPDB	Community Profile Database
CSB	concentrate storage building
CSM	conceptual site model
DEC	Alaska Department of Environmental Conservation
DFG	Alaska Department of Fish and Game
DHSS	Alaska Department of Health and Social Services
DMTS	DeLong Mountain Regional Transportation System
EPA	U.S. Environmental Protection Agency
ERA	ecological risk assessment
ERL	effects range-low
ERM	effects range-median
ESA	Endangered Species Act
ESOD	erythrocyte superoxide dismutase
FWS	U.S. Fish and Wildlife Service
GSD	geometric standard deviation
HHRA	human health risk assessment
IEUBK	integrated exposure uptake/biokinetic
LOAEL	lowest-observed-adverse-effect level
MVUE	minimum-variance unbiased estimate
NAAQS	National Ambient Air Quality Standards
NANA	NANA Regional Corporation
NEC	no-effect concentration
NHANES	National Health and Nutrition Examination Survey
NOAEL	no-observed-adverse-effect level
NPDES	National Pollutant Discharge Elimination System
NPS	National Park Service
NTP	National Toxicology Program
ORNL	Oak Ridge National Laboratory
PEC	probable effect concentration
PRG	preliminary remediation goal
RBC	risk-based concentration
RDA	recommended daily allowance
RfD	reference dose
RME	reasonable maximum exposure
SQS	sediment quality standards
TEC	threshold effect concentration

Teck Cominco	Teck Cominco Alaska Incorporated
THQ	target hazard quotient
TRV	toxicity reference value
UCL	upper confidence limit
USGS	U.S. Geological Survey
WDOE	Washington State Department of Ecology

Foreword

Purpose of the Risk Assessment

Elevated metals concentrations have been identified in tundra in areas surrounding the DeLong Mountain Regional Transportation System (DMTS), primarily as a result of deposition of fugitive dust originating from the DMTS corridor that is used to transport zinc and lead ore concentrates from the Red Dog Mine, which is operated by Teck Cominco Alaska Incorporated. The purpose of the DMTS fugitive dust risk assessment is to estimate possible risks to human and ecological receptors posed by current and future exposure to metals in soil, water, sediments, and biota in areas surrounding the DMTS. The risk assessment is part of the overall process in which the areas of fugitive dust deposition surrounding the DMTS are being evaluated (see the main text for a review of regulatory context). The results of the risk assessment will help risk managers to determine what additional actions may be necessary to reduce those risks.

What This Document Includes

This document is a revised draft of the risk assessment work plan previously produced in January 2003 (Exponent 2003b). The work plan was revised based on comments from individuals (e.g., village residents), non-governmental organizations (e.g., Trustees for Alaska, NANA Regional Corporation, Alaska Industrial Development and Export Authority), and government agencies (e.g., Alaska Department of Environmental Conservation [DEC], National Park Service).

The work plan identifies and describes the tasks necessary to complete the risk assessment, and also summarizes the risk assessment work conducted to date. Preliminary human health and ecological conceptual site models are presented and then refined based on the results of the screening of chemicals of potential concern (CoPCs). Methods to be used for human health and ecological risk calculations are presented, and methods for calculation of risk-based cleanup levels are also discussed. Additional data needs are identified based on the refined conceptual site models, identified CoPCs, and the risk calculations that need to be conducted to compete the risk assessment.

Where We Are in the Process, and What Comes Next

Based on the data needs identified in this work plan, a field sampling plan will be developed to guide data collection in the summer 2004 field season. After those data are available, the risk assessment work will be completed as described in the work plan. A draft risk assessment report is expected to be completed in the first quarter of 2005, at which time DEC will provide a public comment period. After comments are provided, the risk assessment will be finalized, and DEC will determine what actions may be needed to address any risks that are identified.

1 Introduction

The purpose of the DeLong Mountain Regional Transportation System (DMTS) fugitive dust risk assessment is to estimate the magnitude and probability of unacceptable risks to human and ecological receptors posed by current or future exposure to metals in soil, water, sediments, and biota in areas surrounding the DMTS.[1] Elevated metals concentrations have been identified in tundra in areas surrounding the DMTS, primarily as a result of deposition of fugitive dust originating from the DMTS corridor.[2]

This work plan identifies and describes the tasks necessary to conduct the assessment. The risk assessment is being conducted under 18 AAC 75.340(f) as a Method Four cleanup. As such, the risk assessment is part of the overall process in which the areas of fugitive dust deposition surrounding the DMTS are being evaluated under the "site cleanup rules" in the Alaska Administrative Code, sections 18 AAC 75.325 through 75.390, and in accordance with the Alaska Department of Environmental Conservation (DEC) risk assessment procedures manual (DEC 2000) and the decision-making framework illustrated in Figure 1-1, from DEC et al. (2002). The results of the risk assessment will help risk managers to determine what additional actions may be necessary to reduce those risks. In the interim (while the risk assessment is being performed), a number of actions have been and are being taken by Teck Cominco Alaska Incorporated (Teck Cominco) to minimize future fugitive dust generation, and to recover and recycle material containing ore concentrates.

1.1 Site Overview

The Red Dog Mine is located approximately 50 miles east of the Chukchi Sea, in the western end of the Brooks Range of Northern Alaska (Figure 1-2). Base metal mineralization occurs naturally throughout much of the western Brooks Range (Figures 1-3 and 1-4), and strongly elevated zinc, lead, and silver concentrations have been identified in many areas (DEC et al. 2002). The mine is located on land owned by the NANA Regional Corporation (NANA; see

land ownership and use map, Figure 1-5). The geographical area for the risk assessment is the DMTS corridor extending from the Red Dog Mine to the port, including the road, the port facilities, outlying tundra areas, and the marine environment at the port; as well as the area outside of the solid waste permit boundary around the mine. The mine area within the solid waste permit boundary (shown in Figure 1-5) will not be addressed in the risk assessment.

The Red Dog Mine operations began in 1989. Ore containing lead sulfide and zinc sulfide is mined and milled to produce lead and zinc concentrates in a powder form. These concentrates are hauled year-round from the mine via the DMTS road to concentrate storage buildings

[1] In this document, "the DMTS" is used to refer to the entire transportation corridor from the mine to the deepwater ships, including the road, the port facilities, and the barges.

[2] "Fugitive dust" is defined herein as any dust or particulate matter that is emitted to the ambient air from operational activities. Along the DMTS corridor, fugitive dust may be ore concentrate, road dust, or a combination of both.

(CSBs) at the port, where they are stored for later loading onto ships during the summer months. The storage capacity allows mine operations to proceed year-round. During the shipping season, the concentrates from the storage buildings are loaded into an enclosed conveyor system and transferred to the shiploader, and then into barges (Figure 1-6). The barges have built-in and enclosed conveyors that are used to transfer the concentrates to the holds of deepwater ships.

A moss study performed in 2000 by the National Park Service (NPS) (Ford and Hasselbach 2001) found elevated concentrations of metals in tundra along the DMTS road and near the port, apparently resulting from fugitive dust from these facilities. A fugitive dust study completed by Teck Cominco in 2001 (Exponent 2002a) provided an initial characterization of the nature and extent of fugitive dust releases from the DMTS corridor and provided baseline data from which to monitor the performance of new transport and handling equipment and dust management practices. Additional characterization was completed by Teck Cominco at the port site in 2002 (Exponent 2003b, Teck Cominco 2003). A sampling program designed to support the risk assessment was conducted in 2003 to obtain data for additional analytes in multiple environments and media. This program is described in the field sampling plan (Exponent 2003e), and in Appendix A of this document. Appendix B provides a data quality review for the 2003 field program.

1.2 Document Organization

This risk assessment work plan is organized in the format that is anticipated for the risk assessment document. Each section is completed to the extent possible at this stage of the assessment, and describes the work to be completed for the assessment. The sections of the document include:

- Section 1, *Introduction*
- Section 2, *Preliminary Conceptual Site Model*
- Section 3, *Selection of Chemicals of Potential Concern*
- Section 4, *Human Health Risk Assessment*
- Section 5, *Ecological Risk Assessment*
- Section 6, *Calculation of Risk-Based Cleanup Levels*
- Section 7, *Conclusions and Data Needs*
- Section 8, *References*.

Appendices include:

- Appendix A, *Summary of Phase I Sampling Program for the DMTS Fugitive Dust Risk Assessment*
- Appendix B, *Data Quality Review for Phase I Sampling Program for the DMTS Fugitive Dust Risk Assessment*
- Appendix C, *Inorganic Chemical Data Tables*
- Appendix D, *Organic Chemical Data Tables*

2 Preliminary Conceptual Site Model

A conceptual site model (CSM) is a planning tool used for identifying chemical sources, complete exposure pathways, and potential receptors on which to focus the risk assessment. The CSM describes the network of relationships between chemicals released from a site and the receptors that may be exposed to the chemicals through pathways such as ingestion of food or water. The CSM examines the range of potential exposure pathways and identifies those that are present and may be important for human and ecological receptors, and eliminates those pathways that are incomplete and therefore do not pose a risk.

The preliminary CSM for the Red Dog fugitive dust risk assessment describes possible sources and transport mechanisms of metals from the DMTS corridor into surrounding terrestrial and aquatic ecosystems, and the pathways by which receptors may be exposed to those metals. It was developed based on site history, site conditions, and the results of available site sample analyses.

The following sections identify chemical sources and transport mechanisms, as well as the preliminary human health and ecological CSMs. Refined human health and ecological CSMs are presented later in the document, following screening of chemicals of potential concern (CoPCs).

2.1 Sources of Chemicals

The primary chemicals of interest are the ore concentrate components described below. Also reviewed below are other chemicals that have been released in spills of non-metal materials.

2.1.1 Ore Concentrates

The sources of metals associated with the DMTS are the lead and zinc ore concentrates that are produced at the mine; transported over the DMTS road in trucks; and stored, handled, and loaded at the DMTS port facility. Typical concentrations of constituents in the lead and zinc concentrates are illustrated in Table 2-1. In this document, the terms “metals” and “chemicals” are both used to refer to the components of the ore concentrates. Although some components are non-metals or metalloids, most of the constituents of interest are metals

2.1.2 Petroleum Hydrocarbons and Other Spills

There have been historical spills of non-metal materials along the DMTS corridor over the years of operation. DEC provided a list of spills from the Prevention and Emergency Response and Preparation database, which includes spills from 1995 to the present, but not earlier (with the exception of one significant diesel spill that occurred in 1993, which was included on the list). DEC’s list was sorted into DMTS-related spills and mine-related spills, and compared with available records to clarify spill information (Hagy 2003, pers. comm.). A list of DMTS-related

spills from DEC's database is provided in Table 2-2. According to the DEC database, the spills include diesel, engine oil, hydraulic oil, lead concentrate, zinc concentrate, and "other."

There have been a number of small diesel spills (from 10 gallons to 70 gallons) resulting from overfilling trucks at the truck fill station at the port, and one large diesel spill (original estimate of 36,000 gallons, later estimated to be approximately 22,000 gallons) from Fuel Storage Tank #2 (Tank 2) at the port site. The truck fill station has been paved with a concrete apron that drains to a sump, from which the liquid is collected and processed at the mine. The smaller diesel spills listed in Table 2-2 were cleaned up at the time of the spill, and are recorded as cleaned up in the DEC spill database (right column of Table 2-2). Although the diesel spills are recorded as cleaned up, it was unclear what the final concentrations were at the time that DEC issued a "No Further Action" letter. Therefore, sample collection was conducted at the former Tank 2 spill area as part of the 2003 field sampling program (Exponent 2003e and Appendix A of this document).

The hydraulic oil spills were typically the result of a failed hose or fitting. According to the DEC data presented in Table 2-2, the volumes of these spills ranged from 10 gallons to 90 gallons. These spills and the engine oil spills were typically cleaned up at the time of the incident, and the spill database shows them as cleaned up, with the exception of one 20-gallon hydraulic oil spill (Table 2-2). This is likely a recordkeeping error, because these small spills were typically cleaned up immediately (Kulas 2004, pers. comm.). Due to the nature and generally small volume of these spills, their prompt cleanup, and the difficulty of identifying their exact location, no sampling was planned for these spills. No PCB-containing oils have been used at the site; the mining operations were begun relatively recently, in 1989 (Kulas 2003, pers. comm.).

A number of DMTS-related spills were marked as "other" in the DEC database. A review of these spills against available records resulted in further clarification of the material spilled (Hagy 2003, pers. comm.). Several spills that had been marked as "other" in the DEC database were determined to be zinc concentrate spills, and were marked as such in Table 2-2 (i.e., Spill No. 96389915901 on 6/7/1996; Spill No. 97389923301 on 8/21/1997; Spill No. 98389932501 on 11/21/1998; and Spill No. 98389903801 on 2/7/1998). One spill marked as "other" was determined to be a lead concentrate spill, and was marked as such in Table 2-2 (i.e., Spill No. 98389921301 on 8/1/1998). Materials in the two remaining spills marked as "other" in Table 2-2 are uncertain, because the information in the DEC database for these two spills is limited, and does not match the information in Teck Cominco's records. One spill is shown in Table 2-2 as 65 gallons of "other" spilled on 10/5/1997. Teck Cominco's records show that 500 gallons of process water were spilled on 10/5/1997 at the mill process water tank (within the mine). The other spill is shown in Table 2-2 as 200 lb of "other." Teck Cominco's records show a spill of 1 gallon of ethylene glycol at the mine. These two spills appear to have occurred within the mine area, which is not part of the area being addressed by the DMTS risk assessment. Another spill that DEC has inquired about was determined to have occurred at the mine (Spill No. 99389906101 on March 2, 1999), and as such was not listed in Table 2-2.

The lead and zinc concentrate truck spills listed in Table 2-2 are a partial list, because a number of the concentrate spills occurred prior to 1995 (spills prior to 1995 are not included in the DEC database). Lead and zinc concentrate truck spills are initially recovered at the time of the spill.

Follow-up characterization, recovery and recycling of material, and closure of these sites is being conducted by Teck Cominco under the requirements of the Settlement Agreement entered in DEC Case No. 00-354-84-214. As such, the ore concentrate truck spills will not be addressed further in the risk assessment. Refer also to the spill site characterization plan (Exponent 2002c), and the concentrate recovery and recycling plan (Exponent 2002d) and addendum (Exponent 2003d). The characterization process was completed in 2003, and recovery and recycling (where necessary) have been initiated. Results will be reported to DEC per the requirements of the Settlement Agreement identified above. In general, the concentrations remaining at the former concentrate spill sites after removal are lower than the concentrations observed in surrounding areas that result from typical transport and deposition mechanisms.

Other chemicals such as milling reagents are also stored, handled, and transported within the DMTS corridor. There are no reported spills of these materials.

2.2 Transport and Fate of Chemicals

Historically, the primary mechanisms by which metals have escaped into the environment are via windblown dust from the port facilities (buildings, conveyors, etc.), by truck tracking (i.e., tracking of concentrate out of loading and unloading facilities on haul truck tires and other truck surfaces and subsequent deposition onto the road), and by concentrate spillage or escapement from haul trucks, followed by windblown transport as fugitive dust. Additionally, runoff from precipitation and snowmelt could also transport metals from the DMTS road and port operations into surrounding ecosystems. Once released to the environment, some of the metals may become dissolved or suspended in surface water, co-deposited with or adsorbed to sediments, incorporated into soil, and potentially enter the food web through uptake into plants and animals, which then could be consumed by people or upper trophic-level ecological receptors. The following sections briefly describe fugitive dust metal sources and current and past primary transport mechanisms related to these sources.

2.2.1 Road

A number of potential sources of metals and current and past transport mechanisms associated with the DMTS road have been identified. These include:

- **Road construction and maintenance materials—**Road construction and maintenance materials include the materials originally used to construct the road, gravel used for ongoing road repair, and surface water applied regularly to keep down dust on the road. Core samples have shown that elevated metals occurrences on the road are a surface phenomenon, and are not likely associated with the materials originally used to construct the road or regularly added to the crushed base during maintenance (Exponent 2002a). Samples from the gravel and road water source sites confirmed that these materials are an insignificant source of metals to the DMTS road (Exponent 2002a).

- **Tracking along the DMTS road—**Ore concentrate can be tracked out of loading and unloading facilities on haul truck tires and other truck surfaces

and subsequently deposited onto the road. This appears to have been one of the primary sources and release mechanisms over the life of the operation. Recent measures, described in Section 2.2.4, have lessened this transport mechanism.

- **Concentrate spillage and escapement from haul trucks—**Historically, this has included leakage from side doors or blowing from under the tarp covers on the trucks formerly used during normal transit, or spillage from overturned trailers following accidents. Recent measures, described in Section 2.2.4, have reduced these sources and transport mechanisms.

Transport mechanisms for metals that have been deposited onto the DMTS road or tundra include:

- **Mechanical or wind generated dust from road or tundra surfaces—** Airborne transport of dust generated from road surfaces is likely one of the primary mechanisms by which metals have historically been deposited onto the tundra adjacent to the road. Recent measures, described in Section 2.2.4, have lessened this transport mechanism. In addition, dust could potentially be blown from tundra surfaces (e.g., from tundra along the road) where it had previously been deposited.

- **Surface water runoff from road and tundra surfaces—**Surface water runoff from precipitation and from use of water on the road to help keep dust down may transport metals off the road bed. This mechanism may be important in the immediate shoulder area of the road, but it is not likely to carry dust a long distance compared to airborne transport of dust. In addition, dust may be transported by runoff into streams at road crossings or from the tundra into streams, and could subsequently be carried downstream in water or sediment. The transport of metals to streams may be inhibited by physical filtration within the tundra, or by interactions with organic material in the tundra.

2.2.2 Port

The following list includes a number of potential sources of metals and current and past transport mechanisms associated with port operations. Recent measures, described in Section 2.2.4, have significantly reduced many of these sources and transport mechanisms.

- **Windblown dust from the truck unloading building and CSBs—**When doors to these buildings are opened, wind can carry dust from the buildings into the environment around the port site. Improvements to operational procedures at the CSBs, and modifications to the truck unloading building, described in Section 2.2.4, have significantly reduced these sources.

- **Concentrate spillage and dust leakage from conveyers and surge bin—** Likely a primary source in the past but less significant now due to facility upgrades.
- **Spillage and windblown dust during barge loading—**Although historically there may have been some emissions during shiploading, this source has been significantly reduced by improvements made to the shiploader conveyor in 2003, as described in Section 2.2.4.
- **Spillage and windblown dust from barges during transport—**Not likely a significant source either historically or at present because the concentrate is covered by fixed tarps and undisturbed.
- **Spillage and windblown dust during transfer from barges to deepwater ship—**Dust may emanate from the open slot in the fixed tarp, from the conveyor, or from the open hold of the ship. However, the barge conveyor systems were upgraded in 2003, as described in Section 2.2.4, thereby reducing this source.
- **Spillage and windblown dust from the deepwater ship—**Once the concentrate is within the hold of the deepwater ship, the hatches are sealed shut, and the potential is low for spillage or generation of windblown fugitive dust.

Transport mechanisms for metals that have been deposited onto road surfaces at the port site are similar to those mechanisms described above for the DMTS road. In addition, transport mechanisms for metals-containing dust that has been deposited on soil or tundra at the port site include:

- **Mechanical or wind generated dust from soil or tundra surfaces—**This mechanism is similar to the transport of dust from the DMTS road surface or tundra.
- **Surface water runoff from soil or tundra surfaces—**May be important in the immediate area of the port facilities, but is not likely to carry dust a long distance compared to airborne transport of dust. The transport of metals to streams may be inhibited by physical filtration within the tundra, or by interactions with organic material in the tundra. This mechanism is also limited in part by the collection and treatment of surface water from the CSB area prior to discharge to the Chukchi Sea under a National Pollutant Discharge Elimination System (NPDES) permit.

2.2.3 Mine Vicinity

In the area outside of the mine solid waste permit boundary (Figure 1-5), fugitive dust can be transported from either the mine area or the DMTS and deposited on the tundra. Transport mechanisms for metals-containing dust that has been deposited on tundra include:

- **Mechanical or wind generated dust from tundra surfaces—**Dust could potentially be blown from tundra surfaces where it had previously been deposited.
- **Surface water runoff from tundra surfaces—**Dust deposited on the tundra could be carried into streams, and could subsequently be carried downstream in water or sediment. The transport of metals to streams may be inhibited by physical filtration within the tundra, or by interactions with organic material in the tundra.

2.2.4 Fugitive Dust Control Measures

The fugitive dust transport mechanisms described above have been subject to changes resulting from ongoing efforts to reduce emissions. These changes include the use of newer trucks, implementation of truck washing, test paving of the road near the port, significant upgrades to the surge bin and truck unloading facilities, and full enclosure of the conveyers between the surge bin and the CSBs. In addition, significant modifications were made in 2003 to the barges and the shiploader, including full enclosure of the shiploader conveyor, and installation and upgrade of baghouses to actively collect dust within the barge conveyor system. Truck tracking has been reduced by improved dust control in the unloading building, and by truck washing in the summer. Concentrate spillage and escapement is now limited by newer trucks that produce less dust when unloading, have better handling characteristics to reduce the likelihood of roll over, and have hydraulically closed steel covers and solid sides to prevent concentrate from escaping during normal transit or in the event of an accident. Efforts to minimize transport mechanisms from the DMTS road surface include controls implemented to limit tracking, as well as recovery and recycling of metals-containing road material, and placement of hard surface (pavement) at the port and the first 5 miles of DMTS road. Pavement may help to reduce the amount of mud picked up by the trucks, and may thereby help to reduce the tracking of metals-containing material along the road. Improved dust control procedures have been instituted within the CSBs to reduce fugitive dust emissions during unloading and handling of the concentrates, and the conveyors and surge bin have been upgraded to reduce concentrate spillage and dust leakage from these facilities. The shiploader conveyor and the conveyer on the barge have also been upgraded with more complete enclosure and dust control systems. Ongoing efforts to reduce fugitive dust emissions are described in more detail in the background document (DEC et al. 2002).

Fugitive dust control improvements have also been made in the mine area. These include a procedural change to keep the water in the tailings impoundment at a higher level, such that tailings impoundment sediments remain covered by water, thereby eliminating dust from windblown sediments. Other possible control measures are currently being evaluated.

2.3 Preliminary Human Health Conceptual Site Model

This section describes the preliminary CSM for potential human exposures related to DMTS fugitive dust (Figure 2-1). A CSM is a planning tool used for identifying chemical sources, complete exposure pathways, and potential receptors on which to focus the risk assessment.

The preliminary CSM reflects an understanding of the site prior to a more in-depth analysis of environmental chemical concentrations and prior to screening for CoPCs. The purpose of this step is to ensure that all potential pathways are considered regardless of whether those pathways are complete. An exposure pathway is the course a chemical takes from a source to an exposed receptor. Exposure pathways consist of the following four elements: 1) a source; 2) a mechanism of release, retention, or transport of a chemical to a given medium (e.g., air, water, soil); 3) a point of receptor (human or ecological) contact with the medium (i.e., exposure point); and 4) a route of exposure at the point of contact (e.g., incidental ingestion, dermal contact). If any of these elements are missing, the pathway is considered incomplete (i.e., it does not present a means of exposure). Only those exposure pathways judged to be potentially complete are of concern for human exposure. A refined human health CSM is presented in Section 4 after screening procedures are complete.

As discussed above, a human health CSM describes the ways in which people could potentially be exposed to site-related chemicals. More specifically, the CSM provides information about source(s) of chemicals associated with the site, the ways that the chemicals could move through the environment (i.e., transport and fate), the environmental setting of the site as it relates to human activities, the types of human activity that could result in exposure to site-related chemicals (i.e., receptors), and the ways that people could potentially be exposed to those chemicals (i.e., exposure pathways). Chemical sources and transport and fate are discussed above in Sections 2.1 and 2.2, respectively. Environmental setting, receptors, and exposure pathways are discussed in the following sections.

2.3.1 Environmental Setting

The relevant issues specific to human health exposures at the site are the site setting, land ownership, and land use, all of which help dictate the types of activities that people could engage in on or near the site. The site setting is discussed in Section 1. The background document (DEC et al. 2002) provided a detailed description of land ownership, management, and use in the vicinity of Red Dog Mine and the DMTS road and port. These issues are discussed briefly below and illustrated in Figure 1-5. Groundwater considerations are also summarized below.

2.3.1.1 Land Ownership and Management

Red Dog Mine is located on NANA land (Figure 1-5), and is operated by Teck Cominco. NANA also owns the land in the port area, and leases it to the Alaska Industrial Development and Export Authority (AIDEA). AIDEA owns the DMTS, which includes the port on the Chukchi Sea and the 52-mile road linking the mine and the port. Teck Cominco has a priority and non-exclusive contract to use the road and port for exporting its zinc and lead concentrates, and is responsible for its operation and maintenance.[3] The DMTS road runs through lands owned by the State of Alaska, NANA, and the federally owned Cape Krusenstern National Monument, which is administered by NPS. NANA traded lands it received under the Alaska

[3] There are currently no other users of the road, however, other parties wishing to use the DMTS would need to meet regulatory requirements and have an agreement with AIDEA to finance any necessary capacity increase of the infrastructure.

Native Claims Settlement Act with lands managed by NPS to arrive at an agreement allowing for Congressional action in establishing a corridor through the Monument. U.S. Congress granted a 99-year easement to NANA for the corridor through the Monument.

Under the 1982 agreement with NANA, Teck Cominco financed, constructed, and has been operating the mine and mill, in addition to marketing the concentrates produced. Teck Cominco also has responsibility for employing and training NANA shareholders to staff the operations and the responsibility to protect the subsistence lifestyle of the people in the region. At present, 54 percent of the workers and contractors employed by Teck Cominco are NANA shareholders. Continued educational commitments by NANA and Teck Cominco to the NANA shareholders of the region should enable the companies to someday offer 100 percent native employment at the operation, as outlined in the agreement.

2.3.1.2 Land Uses

There are three primary land uses under consideration in the human health risk assessment (HHRA). These include:

- **Commercial and industrial uses—**The transportation corridor, including the road and port, is currently used for commercial/industrial purposes and such uses are likely to continue in the future. The mine is also an industrial use area, but is not considered in this assessment.
- **Subsistence hunting and gathering—**Subsistence hunting and gathering is very important to the economic, nutritional, and spiritual well-being of northwest Alaskan residents. Approximately one-third of local households are dependent on subsistence hunting and gathering, and 55 percent of these households obtain more than half of their food supply by hunting, fishing, and gathering (U.S. EPA 1984). Subsistence hunting and gathering occurs near the transportation corridor, which is part of the larger subsistence area. Subsistence hunting and gathering is also widely practiced within marine areas, including areas near the port site. Subsistence uses are expected to continue in the future.[4]
- **Residential land use—**There is no residential land use along the transportation corridor, nor is such use expected in the future. However, the potential for fugitive dust to indirectly affect residents of downwind/downstream villages (i.e., Kivalina) will be evaluated. In addition, individuals of all ages are assumed to be able to access soils and subsistence resources along the DMTS. However, this type of exposure will be evaluated as part of the subsistence hunting and gathering land use exposure pathways.

[4] There is a public access plan associated with the ambient air permits for the DMTS road and port, which includes signage and other measures to prevent access within areas that could exceed the national ambient air quality standards. Despite the public access controls, hypothetical usage of these areas will be assumed for the risk assessment work and for screening steps described here.

Recreational use of the area is also possible. Recreational activities that are usually undertaken in the DMTS area include hiking, flying, boating, hunting, fishing, and winter sports (e.g., snowmobiling). However, much of this activity occurs during the subsistence use of the area by local residents. Recreational activities by non-residents are limited because of the restricted and costly access to the area. Therefore, the primary land uses of the transportation corridor that could result in exposure to fugitive dust are subsistence hunting and gathering and commercial and industrial uses and these are the focus of the HHRA.

2.3.1.3 Groundwater Considerations

A permanent subsurface groundwater zone is not expected to exist in the area under consideration due to the presence of an active layer of permafrost. The active layer of the permafrost that underlies this region is usually less than 3 ft thick, but thawing at greater depths can occur beneath large rivers (USGS 2001). The drinking water for the areas under consideration comes from surface water resources.

2.3.2 Potential Receptors

There is potential for people to come into contact with metals transported by fugitive dust, either directly or indirectly. Three groups of human receptors have been identified for the site: workers within the DMTS road and port areas, subsistence hunters and gatherers who may use areas in the vicinity of the road as part of their harvest area, and residents of Kivalina and Noatak to the extent that these villages may be affected indirectly by airborne deposition. Although there is some regional recreational use, any exposure for recreational visitors would be much more limited than for subsistence hunting and gathering in the area.

2.3.2.1 Workers

Workers within the DMTS road and port, and at the mine, can be exposed to CoPCs in several ways. They may be exposed in the workplace and through consumption of subsistence foods and water when they are in the village. These potential exposures are discussed below.

Workplace Exposure—Workers who maintain the road and those with primary responsibilities within the port or the mine have the potential for exposure to metals. Mine or port workers who work directly with ore or ore concentrates would be expected to have the highest potential for exposure to metals based on the concentrations in these materials and the higher frequency of potential contacts. Workplace exposure is controlled through a closely monitored industrial hygiene program, including the use of personal protective equipment, blood lead monitoring, and urine cadmium monitoring. The biomonitoring program covers all employees, including process area workers, administrative staff, and other non-process area workers. These workplace controls provide assurance that safe exposure levels are maintained for mine and port workers. Moreover, the industrial activities are not the subject of this assessment, which is focused on the DMTS corridor and the area peripheral to the mine solid waste permit boundary.

In order to evaluate a worker scenario for the DMTS transportation corridor, a hypothetical worker will be evaluated. This scenario will consider exposure to soil and dust based on concentration data for current conditions along the DMTS corridor.

Workers' Subsistence Exposure—Current and future workers could also be exposed to metals through consumption of locally gathered foods and through contact with environmental media while hunting or harvesting foods, and this pathway will be evaluated in the risk assessment. Workers would not be considered likely to consume as much subsistence foods as individuals who engage in a subsistence lifestyle full-time.

Workers' Cumulative Exposure—The risk assessment will include an estimate of cumulative risk to workers through the evaluation of a hypothetical worker exposed to fugitive dust along the DMTS transportation corridor, as well as exposure through the subsistence pathway (i.e., consumption of subsistence foods and contact with environmental media during hunting and harvesting). For lead exposure, the receptor would be a hypothetical female worker who comes in contact with lead in site media during pregnancy. The adult lead model is designed to address potential effects on the fetus following exposure during gestation. This is a conservative approach because the greatest sensitivity to lead occurs during fetal development, and early childhood. In the risk assessment, the most recent baseline blood lead data from the National Health and Nutrition Examination Survey (NHANES) will be used, as summarized by U.S. EPA (2002a), and then run in the model to evaluate lead hazards related to additional exposures 1) to lead in fugitive dust during work on the transportation corridor, 2) from consumption of subsistence foods, and 3) from environmental exposures while hunting and harvesting. The same exposure pathways will be evaluated for non-lead metals, but using standard DEC and U.S. Environmental Protection Agency (EPA) risk assessment methodology, as described in Section 4.2, with input parameters appropriate to an adult worker's potential exposure.

This approach will provide an assessment of cumulative worker/subsistence user exposure to CoPCs that are not assessed under the biomonitoring program. In addition, it will provide an additional measure of health protection by assessing lead and cadmium exposure using more conservative environmental standards (relative to workplace standards).

2.3.2.2 Subsistence Hunters and Gatherers

The subsistence group includes people who fish, hunt, and gather plants and berries, and other family or community members who share those foods. As described in Section 2.3.3, most of the primary exposure pathways evaluated in the risk assessment will focus on this group.

2.3.2.3 Residents

The closest villages to the DMTS road and port are Kivalina and Noatak, and thus the residents of these villages are potential receptors. Given the distance between the villages and the DMTS road and port site, fugitive dust is not expected to significantly impact air, soil, or drinking water within the villages. However, because some streams crossing the DMTS flow into the Wulik River, which in turn provides drinking water for Kivalina, surface water will be evaluated as a drinking water source.

Ambient air modeling performed during the air permitting process has demonstrated that air concentrations beyond the ambient air boundaries (see Figure 1-5) do not exceed National Ambient Air Quality Standards (NAAQS).[5] However, one year of air monitoring is planned for both Kivalina and Noatak, partly in response to community concern. Lead data will be collected and air concentrations of lead will be compared with NAAQS (Hall 2002, pers. comm.). If concentrations exceed NAAQS, metals concentrations within the village(s) will be further investigated independent of the DMTS risk assessment.

2.3.3 Potential Exposure Pathways

An exposure pathway is the course a CoPC takes from a source to an exposed receptor. As discussed above, exposure pathways consist of the following four elements: 1) a source; 2) a mechanism of release, retention, or transport of a CoPC to a given medium (e.g., air, water, soil); 3) a point of human contact with the medium (i.e., exposure point); and 4) a route of exposure at the point of contact (e.g., incidental ingestion, dermal contact). If any of these elements are missing, the pathway is considered incomplete (i.e., it does not present a means of exposure). Only those exposure pathways judged to be potentially complete are of concern for human exposure.

The potentially complete exposure pathways can be further described as "primary" or "secondary" pathways. Primary pathways are those expected to be major contributors to risk estimates, or pathways of particular community concern. Risks from these pathways will be quantified in the HHRA. Secondary exposure pathways are those not expected to contribute significantly to risk estimates. Secondary pathways will be assessed qualitatively or semi-quantitatively in the risk assessment. Figure 2-1 summarizes the exposure pathways identified at the site based on a preliminary understanding of site conditions. The preliminary CSM will be further refined following screening for CoPCs.

The potential exposure pathways can be categorized under three environments: terrestrial, freshwater, and lagoon and coastal marine. In each of these environments, there may be some potential for exposure to metals through consumption of subsistence foods (e.g., plants, fish, and/or other animals), incidental ingestion or dermal contact with soil/sediment, or ingestion or dermal contact with water.

Based on the information gathered in public meetings in Kivalina and Noatak in June and July 2002 (Sundet 2002a,b, pers. comm.), and consultations with DEC, the following list was developed as being representative of the subsistence foods of importance for human consumption in the area:

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[5] Ambient air boundaries are boundaries established around the perimeter of a facility, and are intended to protect public health and welfare through ambient air quality standards. This boundary determines where air quality needs to be evaluated against the NAAQS using computer dispersion models. Operational areas within the facility boundary/ambient air boundary are protected and regulated by occupational health and safety standards. Dispersion modeling required under the air permits for Red Dog Mine has demonstrated that ambient air quality standards are met at the ambient air boundaries. The ambient air boundaries for the port and mine are shown along with the land ownership and usage in Figure 1-4. The ambient air boundary for the road is located 300 ft on either side of the road centerline.

- Plants: berries, sourdock
- Mammals: caribou
- Birds: ptarmigan
- Freshwater fish: various, based on available data
- Lagoon and coastal marine species: to be evaluated quantitatively if metals concentrations in marine sediment and water are elevated (see discussion in Section 2.3.3.3).

The plants and animals selected represent a range of environmental exposure patterns. Subsistence food consumption of these plant and animal species is described in Section 4.2.

Exposure pathways and receptors are described in more detail in the following sections, along with a discussion of the relative importance of each pathway.

2.3.3.1 Worker and Subsistence Use in the Terrestrial Environment

Subsistence hunters and gatherers could be exposed to metals taken up by plants or animals downwind of the DMTS road or port site through consumption of subsistence harvest foods. Metals from the DMTS road or port facility that have been transported onto plants or tundra soils could be consumed by animals (e.g., ptarmigan and caribou) that are in turn consumed by people. Subsistence use of animals is considered a primary pathway.

People could also consume plants and berries that have taken up metals from the soil or onto which fugitive dust has been deposited. Preliminary risk calculations conducted by the Alaska Department of Health and Social Services (DHSS), based on the first set of DEC salmonberry metals data, did not suggest elevated risks associated with consumption of berries (DHSS 2001). From this initial evaluation of salmonberries collected north and south of the port site, DHSS concluded that salmonberry metals concentrations "are consistent with typical background levels and do not pose a public health concern" (DHSS 2001). Further berry sampling conducted by DEC and Exponent suggested elevated concentrations of some metals at the port site relative to reference conditions. Subsistence use of plants (e.g., berries and sourdock) is considered a primary exposure pathway.

In addition, people could be exposed to metals more directly through incidental ingestion and dermal contact with soil, or inhalation of airborne particulates from soil. Direct exposure to soil and dust could occur in the workplace and/or during subsistence hunting and harvesting. There is a public access plan associated with the ambient air permits for the DMTS road and port that is designed to prevent access to areas within ambient air boundaries. The plan controls access to these areas by providing public information and education, and posting signage at points of possible public access. Despite the public access controls, hypothetical usage of these areas will be assumed for the risk assessment work. Both dermal contact and inhalation exposure are likely to be limited relative to soil ingestion and other pathways and thus are considered to be secondary pathways. Incidental soil ingestion, however, is considered a primary exposure pathway for subsistence hunters and gatherers and workers. DEC (2003) implicitly

acknowledges the relative importance of ingestion and the limited contributions of inhalation and dermal exposure to metals in soil and dust by calculating cleanup levels only for soil ingestion of metals, not for inhalation or dermal exposure. Furthermore, in its cleanup level guidance, DEC (2002) provides an equation for calculating a cleanup level for soil based on ingestion only, but does not provide guidance, nor direct the user to calculate cleanup levels for dermal or particulate inhalation exposure. Issues related to the relative importance of ingestion, inhalation, and dermal contact of soil and dust are described further below.

Soil Ingestion—Soil ingestion estimates represent soil that reaches the gastrointestinal tract through hand-to-mouth activity and through inhaled particles that are subsequently swallowed. Studies have been conducted using soil minerals as tracers to measure the amount of soil ingested by adults and children (e.g., Stanek and Calabrese 2000). Such studies measure the amount of metals in the body after contact with metals-containing soil and do not segregate the metal uptake by exposure route. These studies form the basis of the soil ingestion estimates recommended by U.S. EPA (1997b) that will be applied in the HHRA. Thus, a separate quantification of dust exposure via passive re-entrainment of soil to air and via skin absorption is unnecessary and duplicative because these pathways are implicitly included in the soil ingestion rates. Quantitation of exposure by soil ingestion will thus include the portion directly ingested, the portion inhaled, and the portion absorbed through the skin.

Inhalation of Particulates from Soil—There is potential for exposure to metals following re-suspension of dust from soil. However, this pathway has only a limited influence on risk estimates for metals in soil. Relatively little inhaled dust passes into the lower respiratory tract and lungs, where absorption could potentially occur. Both chemical and physical properties of the inhaled substance play a role in the biological fate of inhaled particles, but particle size is the most important factor for metals sorbed to dust and soil. Inhaled particles greater than 1 micron (micrometer) in diameter, which make up the majority of soil and dust in most environmental settings, are largely transported into the gastrointestinal tract. In its *Issue Paper on Metal Exposure Assessment*, U.S. EPA (2003b) states that:

> … a substantial fraction of the inhaled particles larger than 1 micron can be expected to be deposited in the upper respiratory tract and subsequently transferred by mucociliary transport to the gastrointestinal tract, where fractional absorption may be very much different from that of particles absorbed from the respiratory tract."

Particle size analysis of soil from the DMTS indicates that 98 percent of soil particles are larger than 1 micron in diameter. Thus, the majority of inhaled dust and soil at the DMTS would be expected to be ingested.

EPA Region 9 calculates risk-based concentrations (RBCs) termed preliminary remediation goals (PRGs) based on conservative assumptions about exposure through inhalation (where an inhalation toxicity value is available), dermal contact, and incidental ingestion (U.S. EPA 2003c). Table 2-3 shows the relative importance of these three potential human exposure pathways for residential soil. The EPA Region 9 PRGs are not meant to provide screening concentrations applicable to the DMTS risk assessment. Rather, they are provided as a means of illustrating the relative contributions of inhalation, dermal contact, and ingestion exposure.

The models used to calculate RBCs for inhalation of particulates from soil are updates of risk assessment methods presented in *Risk Assessment Guidance for Superfund Part B* (U.S. EPA 1991) and are identical to the *Soil Screening Guidance: User's Guide and Technical Background Document* (U.S. EPA 1996a,b). EPA applies conservative assumptions regarding inhalation rates (i.e., 20 m^3 per day for an adult and 10 m^3 per day for a child) over 350 days per year and 30 years and a particulate emissions factor derived by EPA. The EPA Region 9 modeling for this pathway also applies conservative assumptions regarding the amount of emission and deposition of particles onto soil.

As shown in Table 2-3, the RBCs derived for inhalation of particulates from soil are 8 to 1,500 times greater than the cumulative RBCs for all the metals except cobalt and chromium(VI) (chromium(VI) typically constitutes a small percentage of the total chromium in soil). As described Section 3.3, soil concentrations for both total chromium and cobalt are consistent with reference conditions. In addition, the maximum site concentrations for total chromium (24 mg/kg) and cobalt (27 mg/kg) are below the inhalation PRGs for residential exposure, which are 30 mg/kg for chromium(VI) and 903 mg/kg for cobalt. Consistent with DEC screening levels (as described in Section 3.3), these PRGs were derived assuming target cancer risk levels of 1×10^{-6}. Moreover, the PRGs are based on residential exposure, which would be much greater than the types of exposure that are expected to occur at the site at present or in the future. The results of this qualitative evaluation indicate that the inhalation pathway would have a limited influence on risk estimates for soil.

Dermal Contact with Metals in Soil—Dermal contact with metals in soil may also result in additional exposure. However, non-lipophilic compounds such as metals are only minimally absorbed. EPA recognizes this in the draft *Risk Assessment Guidance for Superfund: Volume 1: Human Health Evaluation Manual (Part E, Supplemental Guidance for Dermal Risk Assessment*) (U.S. EPA 2001), which provides dermal absorption information for only two chemicals, arsenic and cadmium. The dermal absorption fraction of 0.03 is identified for arsenic and 0.001 is identified for cadmium, both based on studies by Wester et al. (1992, 1993) in which metals were held in place on the skin of monkeys for 24 hours. The U.S. EPA (2001) recommendation replaces the prior wording in the U.S. EPA (1992a) dermal guidance document, which provided a generic absorption fraction of 0.001 for metals that had no specific data on absorption. U.S. EPA (2001) further states that there is insufficient information to estimate dermal exposure for other metals.

Dermal exposure does not have a large effect on risk estimates for arsenic and cadmium. Consistent with guidance in U.S. EPA (2001), EPA Region 9 calculated PRGs for dermal exposure to arsenic and cadmium. As indicated in Table 2-3, the residential PRGs derived to be protective of dermal exposures were 4 mg/kg for arsenic and 698 mg/kg for cadmium. The residential PRGs for ingestion were 0.4 mg/kg for arsenic and 37 mg/kg for cadmium, exactly the same as for cumulative exposure (i.e., ingestion, inhalation, and dermal exposure). This comparison indicates that the dermal exposure route has minimal influence on the risks related to arsenic and cadmium exposure in soil. Moreover, because food chain pathways will have a more substantial influence on site risks due to higher consumption rates relative to soil, the impact of the dermal contact with the soil pathway on the overall assessment is further reduced.

2.3.3.2 Subsistence and Residential Use in the Freshwater Environment

Although existing water and fish data indicate minimal effects, surface water quality could potentially be impacted by metals from the DMTS road or the port. If surface water quality is affected, fish in the streams may accumulate metals, which could then be consumed by subsistence users. Thus, subsistence fish consumption from the freshwater environment has been identified preliminarily as a primary exposure pathway for subsistence users.

Surface water drainages in the vicinity of the road ultimately flow into the Wulik River or into the Chukchi Sea near the port site (south of Kivalina). The Wulik River is a source of drinking water for Kivalina residents. Sampling of Kivalina drinking water has been conducted on an ongoing basis and has not shown elevated metals concentrations (DHSS 2001). Nevertheless, drinking water consumption from the freshwater environment has been identified preliminarily as a primary exposure pathway for residents.

Surface water data will be compared with reference conditions and with RBCs protective of residential drinking water. If concentrations are elevated over reference conditions and the RBCs, exposure to CoPCs through ingestion of drinking water will be quantified in the risk assessment. Surface water data will also be compared with water quality criteria protective of people consuming water and fish, where available. If concentrations are elevated over reference conditions and the water quality criteria, exposure to CoPCs through fish consumption will be quantified in the risk assessment.

2.3.3.3 Subsistence Use in the Lagoon and Marine Environments

Metals could be transported to the lagoon and marine environments through surface water runoff, fugitive dust deposition, or spillage in the barge transfer operation, and could subsequently be taken up by marine animals that are consumed by people. Containment and treatment of surface water runoff at the port site limits the potential for metals migration via surface water. Recent data for lagoon and marine sediment and water near the port will be evaluated to determine whether there are any elevated concentrations in these media near the port. If concentrations are elevated above sediment quality criteria (e.g., Washington State sediment quality standards [SQS]; WDOE 1995), further analyses will be conducted to evaluate risks related to bioaccumulation into, and human consumption of, lagoon and marine foods. Thus, seafood consumption from the lagoon and marine environments has been identified preliminarily as a primary exposure pathway for subsistence users.

Very little direct human contact with marine sediments and water is expected due to the lack of exposed sediment at the site, and the low water temperature, which precludes direct contact through swimming and wading. Similarly, little if any contact with lagoon sediment and water is expected. Direct contact with lagoons within the port ambient air boundaries (i.e., North and South lagoons) is prohibited. Direct contact with sediment and water in lagoons outside the port ambient air boundary is likely to be very low. As with marine waters, low temperatures in the lagoons preclude direct contact through swimming and wading.

Metals concentrations in lagoon and marine sediment and water near the port will be compared with reference conditions, and with sediment screening levels and water quality criteria based

on consumption of fish to determine whether further quantitative risk estimates will be conducted for the marine environment.

2.4 Preliminary Ecological Conceptual Site Model

This section describes the preliminary CSM for potential ecological exposures related to DMTS fugitive dust (Figure 2-2). A CSM is a planning tool used for identifying chemical sources, complete exposure pathways, and potential receptors on which to focus the risk assessment. The preliminary CSM reflects an understanding of the site prior to a more in-depth analysis of environmental chemical concentrations and prior to screening for CoPCs. The purpose of this step is to ensure that all potential pathways are considered regardless of whether those pathways are complete. The following sections characterize the environmental setting, identify potential exposure pathways and receptors, and define preliminary assessment and measurement endpoints for the ecological risk assessment.

2.4.1 Site Description

The Red Dog study area lies within moderately sloping hills, lowlands, and broad stream valleys between the Chukchi Sea and the Delong Mountains (Figure 2-3). The geography of the region is varied, ranging from the rugged steep peaks and valleys in the DeLong Mountains, to more moderate rolling topography on the Brooks Range foothills and Lisburne Hills, to extensive areas of relatively flat tundra cover between the hills and the coast. An active layer of permafrost, usually less than 3 ft thick, underlies this region, but thawing at greater depths can occur beneath large rivers (Ward and Olson 1980).

The climate in the study area is classified as a cold continental climate (Gough et al. 1988). Near the coast, where the Chukchi Sea has a limited moderating effect on the climate, typical summer temperatures range from 39 to 55ºF (4 to 13ºC) and winter temperatures range from −15 to 5ºF (−26 to −15ºC). Summer temperatures at Red Dog Mine typically fluctuate between 36 and 64ºF (2 and 18ºC), and winter temperatures at the mine are commonly around −20ºF (−29ºC). The mean annual precipitation in the study area is approximately 18 in (45 cm), and more than one-half the annual precipitation occurs as rain from July through September; August is the wettest month. Snowfall has been recorded in every month of the year, but consistent snow cover generally occurs only from the middle of October to the middle of May. In early October, ice will begin to form along the coast; however, high winds and high waves can halt the formation of a solid cover until January (RWJ 1997). The Chukchi Sea is covered in ice from mid-November through May or June.

The two primary drainages in the DeLong Mountains area are the Wulik and Kivalina rivers, which flow to the Chukchi Sea (Figure 2-3). Both of these rivers are located to the north of the DMTS road. To the south and east of the DMTS road corridor lies another major drainage, the Noatak River. With the exception of the Evaingiknuk Creek drainage basin, which flows to the Noatak River, all of the streams crossed by the DMTS road drain to the Wulik River. The tributaries in this area tend to have high flows in the spring due to snow melt and low or no flows in the winter, when most creeks freeze and stop running (Dames & Moore 1983a).

Reaches of Anxiety Ridge Creek and Aufeis Creek are shown in Photographs 1 and 2. Other aquatic and semi-aquatic habitats in the study area include the nearshore marine environment, open and closed coastal lagoons near the port site, temporary and permanent tundra ponds, and marshes, wet meadows, and other wetlands. Port Lagoon North, situated between port facilities to the east and the Chukchi Sea to the west, is shown in Photograph 3. Photographs 4 and 5 show typical tundra ponds found onsite during the summer. Tundra ponds range from small, shallow areas of flooded tundra to larger pools surrounded by dense emergent vegetation.

The vegetation over the study area is classified as mesic graminoid herbaceous (grass and sedge) and dwarf scrub/shrub communities. The mesic graminoid herbaceous communities consist of tussock-forming sedges, such as cottongrass (*Eriophorum* spp.) and stiff sedge (*Carex bigelowii*), mosses, and lichens (USGS 2001). Common dwarf shrubs found in this region include dwarf arctic birch (*Betula nana*), crowberry (*Empetrum nigrum*), narrow-leaf Labrador tea (*Ledum decumbens*), and mountain cranberry (*Vaccinium vitis-idaea*). The dwarf scrub communities are composed of *Dryas* species, prostrate willows (e.g., *Salix reticulata* and *S. phlebophylla*), and ericaceous species (e.g., *Vaccinium* spp., *Cassiope tetragona*, and *Arctostaphylos* spp.). In areas with low scrub vegetation, the most prevalent trees are willows (USGS 2001; Dames & Moore 1983a). Most of the area surrounding the DMTS road corridor is tussock tundra intergraded with low shrub formations, as shown in Photographs 6 and 7. In the port area, lyme grass (*Elymus mollis*) and beach pea (*Lathyrus maritimus* var. *pubescens*) dominate along the sand dunes (Dames & Moore 1983a).

2.4.2 Sensitive Species

Section 7(a) of the Endangered Species Act (ESA; CFR 402) requires federal agencies, in consultation with the U.S. Department of the Interior and National Marine Fisheries Service, as appropriate, to ensure that the actions that they authorize, fund, or carry out are unlikely to jeopardize the continued existence of a threatened or endangered species, or adversely modify or destroy their critical habitat. As required by Section 7 of the ESA, a *Biological Assessment of the Red Dog Mining Project's Potential Effects to Endangered Species* (U.S. EPA 1984) was prepared to complement the environmental impact statement issued in 1984. The biological assessment concluded that the arctic peregrine falcon (*Falco peregrinus tundrius*) was the only listed terrestrial species present in the study. The U.S. Fish and Wildlife Service (FWS) has since delisted the arctic peregrine falcon, but it is currently an Alaska "species of special concern." The biological assessment also identified the endangered bowhead whale (*Balaena mysticetus*) as a seasonal migrant that may occur in the study area during the spring (U.S. EPA 1984).

EPA also conducted a Section 7 consultation when it issued a NPDES permit for the port site. According to the fact sheet for Teck Cominco's NPDES permit for the Red Dog port site (NPDES permit number AK-004064-9), the spectacled eider (*Somateria fischeri*) and the Steller's eider (*Polysticta stelleri*) are threatened species that may occur in the area where treated surface water is discharged. The eiders migrate through the area in the spring and fall. The port site is not a designated critical habitat. FWS determined that no endangered species were likely to occur within the project area of the port site's discharges, but that the bowhead whale and the endangered Steller or northern sea lion (*Eumetopias jubatus*) seasonally occur in

the Chukchi Sea. EPA determined that discharges would not affect these species (NPDES permit number AK-004064-9).

2.4.3 Sensitive Environments

The Alaska Administrative Code (18 AAC 75.990) defines an "environmentally sensitive area" as a geographic area that is particularly susceptible to change or alteration, including rare or vulnerable natural habitats; areas of high natural productivity or essential habitat for wildlife; unique geologic or topographic features that are susceptible to a discharge; floodplains or other areas that protect, maintain, or replenish land or resources; and state and federal protected areas, such as wilderness areas, parks, and wildlife refuges. Several sensitive environments occur in the vicinity of the DMTS road and port site; the most notable is Cape Krusenstern National Monument, which surrounds 24 miles of the DMTS road and the port site (Figure 1-5). The Noatak National Preserve and the Noatak River, a National Wild River, are sensitive environments located east of the DMTS road corridor. (The National Wild and Scenic Rivers Act designates some rivers or river reaches as "wild" or "scenic" or both.) To the north, the Wulik River, Ikalukrok Creek, Imikruk Creek, and the Omikviorok River are designated by the Alaska Department of Fish and Game (DFG) as "waters important for spawning, rearing or migration of anadromous fishes" (DFG 1998). New Heart Creek and Tutak Creek, which cross the DMTS road, also have this designation. Freshwater and saltwater wetlands and land "with continuous natural terrestrial vegetation cover" (AAC 75.630) are other sensitive environments that occur in the study area.

2.4.4 Potential Exposure Pathways

An exposure pathway is the course a chemical takes from a source to an exposed receptor. As discussed previously, exposure pathways consist of the following four elements: 1) a source; 2) a mechanism of release, retention, or transport of a chemical to a given medium (e.g., air, water, soil); 3) a point of contact with the medium (i.e., exposure point); and 4) a route of exposure at the point of contact (e.g., incidental ingestion, dermal contact). If any of these elements are missing, the pathway is considered incomplete (i.e., it does not present a means of exposure). Only those exposure pathways judged to be potentially complete are of concern for ecological receptors. Additionally, exposure to naturally occurring metals is likely throughout the area, both beyond and within the area of the DMTS, through the pathways described above. Exposure to fugitive dust releases represents an incremental exposure above the exposure to naturally occurring metals.

Potential pathways by which ecological receptors may be exposed to metals associated with the DMTS exist for both terrestrial and aquatic communities in the vicinity of the DMTS road and port facility, as illustrated in the preliminary CSM for the DMTS ecological risk assessment (Figure 2-2).

Primary exposure pathways are those expected to contribute highest to risk estimates, while secondary exposure pathways are not expected to increase risk substantially. Primary exposure pathways for terrestrial receptors include the consumption of plant material or prey and the incidental ingestion of soil. For plants, the primary pathways are the uptake of metals

incorporated into soil and the uptake of metals deposited onto plant surfaces as fugitive dust (Figure 2-2). Soil fauna may also be exposed to metals through direct contact with the soil. Primary exposure pathways for aquatic receptors include the ingestion or uptake of surface water, consumption of plant material or prey, incidental ingestion of sediment during foraging, and direct contact with surface water (Figure 2-2). Some aquatic receptors may also be exposed through the uptake of metals from sediments. However, in most situations, dermal contact and inhalation are less important sources of metals exposure in wildlife than food and incidental soil ingestion (Newman et al. 2003). The external epithelium, an effective barrier to inorganic metals, minimizes the dermal uptake of metals in higher organisms (Drexler et al. 2003), and in general, inhalation of particles is assumed to be insignificant compared to other exposure routes for metals and is not addressed in ecological risk assessment (Newman et al. 2003).

2.4.5 Potential Receptors

Potential ecological receptors that may be exposed to metals from the DMTS occur in terrestrial systems such as shrub and tussock tundra, as well as aquatic systems such as creeks near or crossing the DMTS road, tundra ponds, coastal lagoons, and the marine ecosystem. The receptors comprise a wide range of life histories, from small herbivorous mammals that could complete their entire life cycles in small home ranges near the DMTS road, to migratory waterfowl that forage and breed on coastal lagoons during summer months and then migrate. Large-bodied herbivorous and carnivorous mammals that roam widely in search of food may be exposed in multiple areas near the DMTS road and port, but are also likely to forage outside of areas where fugitive dust deposition has occurred. Forage areas both within and beyond the deposition area have naturally occurring metals that contribute to exposure for various receptors.

Categories of ecological receptors that are potentially affected include terrestrial plants, aquatic and wetland plants, soil fauna, aquatic invertebrates, fish, birds, and mammals (Figure 2-2). Each category encompasses a range of functional groups, such as terrestrial plant-eaters (herbivores) or freshwater fish-eaters (piscivores), that differ by habitat utilization and preferred foods. The particular species composition of aquatic and terrestrial communities varies among habitats near the DMTS road and port. Thus, some receptor categories are not represented in all communities near the DMTS road corridor.

2.4.6 Preliminary Assessment and Measurement Endpoints

This section defines preliminary assessment and measurement endpoints and presents the rationale for selection of representative receptors. The preliminary assessment endpoints are components of the ecosystem that represent important environmental values and that may be susceptible to adverse effects from exposure to metals in fugitive dust. The preliminary assessment endpoints identified for the risk assessment are the structure and function of plant, invertebrate, and fish communities and the survival, growth, and reproduction of wildlife populations that inhabit the DMTS road corridor. These endpoints include the following:

- Structure and function of:
  - Terrestrial plant communities
  - Freshwater aquatic and wetland plant communities
  - Marine aquatic and wetland plant communities
  - Soil fauna communities
  - Freshwater aquatic invertebrate communities
  - Freshwater fish communities
  - Marine aquatic invertebrate communities
  - Marine fish communities
- Survival, growth, and reproduction of terrestrial avian:
  - Herbivore populations
  - Invertivore populations
  - Carnivore populations
- Survival, growth, and reproduction of terrestrial mammalian:
  - Herbivore populations
  - Invertivore populations
  - Carnivore populations
- Survival, growth, and reproduction of freshwater avian:
  - Herbivore populations
  - Invertivore populations
  - Piscivore populations
- Survival, growth, and reproduction of freshwater mammalian:
  - Herbivore populations
  - Piscivore populations

- Survival, growth, and reproduction of marine avian:
  - Herbivore populations
  - Invertivore populations
  - Piscivore populations
- Survival, growth, and reproduction of marine mammalian:
  - Invertivore populations
  - Piscivore populations
  - Carnivore populations.

The preliminary measurement endpoints to be used to evaluate the attainment of assessment endpoints such as the structure and function of plant, invertebrate, and fish communities are the range of concentrations of CoPCs measured in soil, sediment, and surface water at the site relative to ecological screening benchmarks. For assessment endpoints such as the survival, growth, and reproduction of various bird and mammal populations, indicator species that are representative of broader functional groups will be used to evaluate ecological risk to those groups. These indicator species, or ecological receptors, were selected taking into consideration a variety of factors, including:

- Occurrence at the site
- Completeness of the exposure pathway
- Sensitivity to contaminant exposure
- Home range size appropriate for evaluating ecological risk across a broad site
- Availability of exposure data
- Societal value.

Whenever possible, species that are harvested for subsistence use were selected as ecological receptors. These species were chosen from subsistence lists developed at public meetings in Kivalina and Noatak in June 2002 (Table 2-4; Sundet 2002a,b, pers. comm.). Where appropriate, receptors were also selected from the *User's Guide for Selection and Application of Default Assessment Endpoints and Indicator Species in Alaskan Ecoregions* (DEC, no date).

The preliminary measurement endpoints for bird and mammal populations are the range of modeled dietary exposures of each representative receptor to CoPCs as compared to toxicity reference values (TRVs) derived from the literature. Preliminary assessment endpoints, measurement endpoints, and representative receptors are summarized in Table 2-5.

3 Selection of Chemicals of Potential Concern

The following sections describe the screening and selection of CoPCs, including a target chemical list, a review of available data, and the human health and ecological CoPC screening and selection sections.

3.1 Target Chemical List

Table 3-1 illustrates the target list of chemicals to be evaluated in the CoPC screening. This list is based on the list of concentrate constituents (Table 2-1) excluding bismuth, calcium, chloride, gallium, germanium, gold, silicon, sulfate, and sulfur. The latter chemicals are not included on the list because:

1. With the exception of calcium, these constituents are not on EPA's target analyte list, nor are they on DEC's list of hazardous substances for which cleanup levels are provided in 18 AAC 75.340 and 18 AAC 75.345. The DEC risk assessment procedures manual (DEC 2000) explains that these lists were developed using the Pareto principal, and goes on to say that: "the Pareto principal … states that a relatively large number of problems (for example, a large proportion of site attributable risk) in a given situation will be found to be caused by only a few factors (or a few hazardous substances). … the target analyte list [substances] … are those manufactured and used in the greatest amounts and that are the most toxic."
2. There are no relevant human health or ecological toxicity criteria for these constituents (because they are generally not considered to be a hazard), and therefore they cannot readily be evaluated.
3. For most of these constituents, data have not been collected.
4. Bismuth, gallium, germanium, and gold occur at relatively low concentrations in the concentrate, and calcium, chloride, silicon, sulfate, and sulfur are naturally abundant in the environment.

Organic compounds associated with former petroleum hydrocarbons are not included on the list because: 1) they occur in very localized areas at former petroleum spill sites, primarily in the Tank #2 area at the port site; and 2) they occur at depth or beneath pavement, and therefore do not have exposure potential. Available data for organic chemicals are attached in Appendix D.

3.2 Review of Existing Soil, Sediment, and Water Data

This section provides an overview of prior data collection, discusses data usability criteria, and reviews soil, tundra soil[6], sediment, and water data that were used in the CoPC screening and that are available for use in the risk assessment. Data are reviewed by environment and medium in the following subsections.

3.2.1 Prior Studies

Table 3-2 provides an overview of prior studies conducted in the Red Dog area. The studies include those led by Teck Cominco and its consultants, and state and federal agencies, between 1978 and the present. Not all of these data are suitable for use in the risk assessment. The following section discusses data usability considerations and criteria.

3.2.2 Data Usability

The studies listed in Table 3-2 have widely varying usability for the CoPC screening and the risk assessment. The criteria used to select data for these analyses are described in this section. These include the following:

- **Year of Collection—**For several reasons, recent datasets were typically used in the CoPC screening. First, conditions change over time, because the environments at the site are dynamic, both in terms of environmental conditions (e.g., climate and weather), and in terms of dust deposition. Thus, the most recent data best represent the current distribution and magnitude of chemical concentrations in media at the site. Second, in many cases, more recent data are available in the same areas or at the same stations where older data were collected. Generally, data collected between 2001 and 2003 were used in the CoPC screening. Older data were used for locations where there has not been more recent data collection. Third, for the most part, older datasets primarily included the analytes lead, zinc, and cadmium, while the more recent datasets (especially the 2003 sampling) include a longer analyte list to facilitate CoPC screening.

- **Sample Depth—**In soil, tundra soil, and sediment, surficial samples (the shallowest sample interval at a given sample station) were used in the assessment, because the fugitive dust deposition is a surface phenomenon, and the most elevated concentrations are typically found in the shallowest depth interval (Exponent 2003c). Also, human and wildlife receptors are most likely to be exposed to soil or tundra soil from the shallowest sample depth interval.

[6] Note that "soil" refers to inorganic soil, principally found on the road and facility areas. "Tundra soil" refers to the peaty organic material immediately beneath the live tundra mat.

- **Paving or Removal—**Soil samples that have been removed by excavation (i.e., for recovery and recycling), or that are covered with pavement, were excluded from the screening, because they no longer represent an exposure medium for human or wildlife receptors.

- **Comparability—**Samples that are not directly comparable were not used in the screening analysis. For example, U.S. Geological Survey (USGS) sediment samples (Brabets 2003, pers. comm.) were sieved to a fine mesh size before analysis, and therefore are not representative of in-place sediment, and are not directly comparable to conventional sediment samples. USGS sediment data were not used in the screening. Also, tundra surface samples collected in the port site area by Teck Cominco (Teck Cominco 2003) were gathered to identify areas for possible recovery and recycling. However, the collection methods for the tundra surface samples were different from the methods used in other surveys to collect tundra soil samples and plant samples (e.g., moss, lichen, willow) required for the risk assessment. Therefore, the tundra surface samples collected by Teck Cominco in 2003 were not used in the screening analysis. However, inorganic surface soil samples from Teck Cominco (2003) were comparable to other surface inorganic soil samples in facility fill areas, and therefore these were used in the CoPC screening analysis.

- **Data Quality Review—**Most of the data used in the CoPC screening and available for use in the risk assessment have been validated and qualified as part of a normal quality assurance review process. The quality assurance review for the 2003 risk assessment data collection program is provided in Appendix B. A few data sets of lesser importance for the risk assessment were not validated. These included some of the stream water data and port site soil data collected in 2003 by Teck Cominco (Teck Cominco 2003). The most important stream water data sets were validated (i.e., the September and October 2003 data sets, for which most or all of the target chemicals were analyzed). Other sets without the full target chemical list (i.e., the months of May through August 2003) were not validated. The Teck Cominco (2003) soil and tundra soil data sets were not validated because there was already significant coverage of these areas with data sets that were previously validated.

Table 3-3 identifies the names of the surveys from which data were used in the CoPC screening, grouped by environment and medium. Citations for the survey sources are also provided in Table 3-3. Table 3-3 shows the sample coverage (number of samples) for site (onsite) and reference (offsite) data that were used in the CoPC screening. Although some of the analytes have a limited number of sample results, the chemicals that have greater sample coverage (i.e., lead, zinc, and cadmium) may be used as indicators for the spatial distributions of the associated chemicals.

Figure 3-1 shows the station locations for soil, tundra soil, sediment, and surface water data; different symbols are used to indicate the types of data that were collected at each station.

Figures 3-2, 3-3, and 3-4 show the sample station locations for soil, sediment, and water, respectively.

Appendix C provides tabulated data by environment and medium. These data were used in the CoPC screening, subject to the criteria described above.

In the following sections, existing soil, sediment, and water data that were used in the CoPC screening and are available for use in the risk assessment will be reviewed by environment and medium, including soil and tundra soil in the terrestrial environment, and sediment and surface water in streams, tundra ponds, lagoons, and the marine environment.

3.2.3 Terrestrial Environment

The following sections discuss media in the terrestrial environment, including site and reference soil and tundra soil. Note that "soil" refers to inorganic soil, principally found on the road and facility areas. "Tundra soil" refers to the peaty organic material immediately beneath the live tundra mat. Figure 3-2 shows the sample station locations. Table 3-3 lists the surveys in which data were collected for these areas, and summarizes the sample coverage by analyte. Data tables are included in Appendix C.

3.2.3.1 Site Soil

Inorganic soil data for the site include samples from road and facility areas (Figure 3-2). The types of samples on the road include road surface and core samples, and road shoulder samples (fine material from the toe of the road embankment). Surface soil sample results are available for the port facility areas.

3.2.3.2 Reference Soil

The reference inorganic soil samples are from material sites that were used to build the DMTS road, and that are used to provide gravel for ongoing maintenance for road and facility areas (Figure 3-2). These samples are representative of the types of geologic materials found in the samples of inorganic soil from site areas (i.e., road and facility areas).

3.2.3.3 Site Tundra Soil

Tundra soil refers to the peaty organic material immediately beneath the live tundra mat. Tundra soil samples have been collected around the port facilities and on transects along the DMTS road (Figure 3-2).

3.2.3.4 Reference Tundra Soil

Reference tundra soil samples were collected from the Phase 1 terrestrial reference area in 2003 (Appendix A). The terrestrial reference area is located to the south of the DMTS, in the prevailing upwind location (Figure 3-2).

3.2.4 Streams

The following sections discuss media in the stream environment, including site and reference stream sediment and surface water. Figures 3-3 and 3-4 show the sample station locations and streams. Table 3-3 lists the surveys in which data were collected for these areas, and summarizes the sample coverage by analyte. Data tables are included in Appendix C.

3.2.4.1 Site Stream Sediment

Sediment data are available for a number of streams along the length of the DMTS road between the mine and the port (Figure 3-3). These include New Heart Creek, Aufeis Creek, Omikviorok River, and Anxiety Ridge Creek. Data are available for multiple stations on each stream, typically at locations some distance upstream and downstream of the road, as well as immediately downstream of the road. For several streams, data are also available for downstream stations prior to confluence with other streams.

3.2.4.2 Reference Stream Sediment

Reference stream sediment samples are available from stations at five streams in the terrestrial reference area (Figure 3-3). The terrestrial reference area is located to the south of the DMTS, in the prevailing upwind location. The streams sampled originate within the reference area.
Site Stream Surface Water

Surface water data are available for a number of streams along the length of the DMTS road between the mine and the port (Figure 3-4). These include New Heart Creek, Aufeis Creek, Straight Creek, Omikviorok River, Mud Lake Creek, Tutak Creek, and Anxiety Ridge Creek. Data are available for multiple stations on each stream, typically at locations some distance upstream and downstream of the road, as well as immediately downstream of the road. For several streams, data are also available for downstream stations prior to confluence with other streams.

3.2.4.3 Reference Stream Surface Water

Reference stream surface water samples are available from stations at three streams in the terrestrial reference area (Figure 3-4). The terrestrial reference area is located to the south of the DMTS, in the prevailing upwind location. The streams sampled originate within the reference area.

3.2.5 Tundra Ponds

The following sections discuss media in the tundra pond environment, including site and reference tundra pond sediment and surface water. Figures 3-3 and 3-4 show the sample station locations, and Table 3-3 summarizes the sample coverage by analyte. Data tables are included in Appendix C.

3.2.5.1 Site Tundra Pond Sediment

Tundra pond sediment samples are available from stations on two transects along the DMTS: one transect at the port site, and one in the middle portion of the road (Figure 3-3).

3.2.5.2 Reference Tundra Pond Sediment

Reference tundra pond sediment samples are available from stations at five tundra ponds in the terrestrial reference area (Figure 3-3). The terrestrial reference area is located to the south of the DMTS, in the prevailing upwind location.

3.2.5.3 Site Tundra Pond Surface Water

Tundra pond surface water samples are available from stations on two transects along the DMTS: one transect at the port site, and one in the middle portion of the road (Figure 3-4).

3.2.5.4 Reference Tundra Pond Surface Water

Reference tundra pond surface water samples are available from stations at three tundra ponds in the terrestrial reference area (Figure 3-4). The terrestrial reference area is located to the south of the DMTS, in the prevailing upwind location.

3.2.6 Lagoons

The following sections discuss media in the lagoon environment, including site and reference lagoon sediment and surface water. Figures 3-3 and 3-4 show the sample station locations. Table 3-3 lists the surveys in which data were collected for these areas, and summarizes the sample coverage by analyte. Data tables are included in Appendix C.

3.2.6.1 Site Lagoon Sediment

Sediment data for the site lagoons include samples at multiple stations in Ipiavik Lagoon, North Lagoon, Port Lagoon North, and Port Lagoon South (Figure 3-3).

3.2.6.2 Reference Lagoon Sediment

Sediment data for the reference lagoons include samples at multiple stations in the Control Lagoon and the Reference Lagoon. The Control Lagoon and Reference Lagoon are located approximately 2 miles and 5 miles, respectively, to the southeast (in the prevailing upwind direction) of the port site facilities (Figure 3-3).

3.2.6.3 Site Lagoon Surface Water

Surface water data for the site lagoons include samples at multiple stations in Ipiavik Lagoon, North Lagoon, Port Lagoon North, and Port Lagoon South. The Control Lagoon and Reference

Lagoon are located approximately 2 miles and 5 miles, respectively, to the southeast (in the prevailing upwind direction) of the port site facilities (Figure 3-4).

3.2.6.4 Reference Lagoon Surface Water

Surface water data for the reference lagoons include samples at multiple stations in the Control Lagoon and the Reference Lagoon. The Control Lagoon and Reference Lagoon are located approximately 2 miles and 5 miles, respectively, to the southeast (in the prevailing upwind direction) of the port site facilities (Figure 3-4).

3.2.7 Marine Environment

The following sections discuss media in the marine environment, including site and reference marine sediment and surface water. Figures 3-3 and 3-4 shows the sample station locations. Table 3-3 lists the surveys in which data were collected for these areas, and summarizes the sample coverage by analyte. Data tables are included in Appendix C.

3.2.7.1 Site Marine Sediment

Marine sediment data for the site (Figure 3-3) include samples from a sampling grid in the nearshore area (located between 0 and 0.25 mile from shore, and up to 0.3 mile to the north and south of the port, centered on the shiploader area). Data are also available for sample stations going out from nearshore areas to offshore areas where deepwater ships are loaded by the lightering barges (approximately 3 miles out) and beyond, out to 6 to 8 miles from shore.

3.2.7.2 Reference Marine Sediment

Reference marine sediment data (Figure 3-3) are available from sample stations approximately 3 miles to the south of the port site (in the prevailing upwind and upcurrent direction).

3.2.7.3 Site Marine Water

Marine surface water data (Figure 3-4) for the site include samples from stations in the nearshore area (located between 0 and 0.25 mile from shore, and up to 0.3 mile to the north and south of the port, centered on the shiploader area).

3.2.7.4 Reference Marine Water

Reference marine surface water data (Figure 3-4) are available from sample stations approximately 3 miles to the south of the port site (in the prevailing upwind and upcurrent direction).

3.2.8 Comparison of Site Data with Reference Data

Comparisons between site and reference area concentrations were conducted using an analysis of variance (ANOVA) model followed by a multiple comparison test. Differences were also assessed using the Wilcoxon rank-sum non-parametric test. Both the Wilcoxon and the multiple comparison tests were one-sided tests for whether the site concentration was significantly greater than the reference. Significance was determined at a 0.05 level (alpha=0.05). The ANOVA method is more powerful than the non-parametric test, but underlying assumptions of equal variance and normality must be met. In cases where the results for parametric and non-parametric test methods did not agree, the underlying assumptions were scrutinized further to determine which method was most reliable for each case. When 50 percent or more of site data values were undetected, statistical analyses were not performed. The results of the statistical comparisons are provided in Tables 3-4 through 3-13. The importance of the site-reference comparisons to the selection of CoPCs varies by analyte, and is discussed below in the CoPC screening and selection sections.

3.2.9 Data Gaps

As shown in Table 3-3, there are a minimum of three analyses for every analyte on the target chemical list (Table 3-2), in each medium and environment, for both site areas and reference areas. There are sufficient data for completion of the CoPC screening in primary media. The results of the CoPC screening analyses will help to identify additional data needs. The most significant data gaps are for biological media. Biota sampling is planned for the summer 2004 field season to fill these data gaps. Biota data needs will be discussed further in Section 7 of this document.

3.3 Human Health CoPC Screening

The human health CoPC screening is used to focus the risk assessment on constituents at the site that have the greatest potential to contribute to human health risks. To ensure that only those constituents that are highly unlikely to contribute even a minimal human health impact are screened out, conservative screening methods are used. The result of the human health CoPC screening is the identification of a site-specific list of chemicals on which the remainder of data evaluation and the risk assessment will be focused. In this investigation, chemicals present in ore concentrates were identified as site-related source materials and are the focus of this screening (Table 3-1; Section 3.1).

The methods used in the CoPC screening are consistent with those described in DEC's *Risk Assessment Procedures Manual* (DEC 2000) and EPA's *Risk Assessment Guidance for Superfund* (U.S. EPA 1989). Specifically, maximum chemical concentrations in each relevant site environmental media were compared with two types of screening levels. First, because the constituents of interest in site source materials are all chemicals that occur naturally in soil, site chemical concentrations were statistically compared to reference concentrations from samples collected in areas not impacted by site activities. The locations from which reference samples were collected and the statistical methods used to compare site and reference samples are described above in Section 3.2. Second, site concentrations were compared to human health-

protective risk-based screening levels (DEC 2003) derived using conservative residential screening levels, and further divided by an additional safety factor of 10 (i.e., representing a cancer risk of 1×10^{-6} or a hazard index of 0.1). For each environmental media, those chemicals that both exceeded their risk-based screening level *and* were significantly different than reference concentrations were retained as human health CoPCs. Site concentrations below screening levels indicate that a risk to human health is highly unlikely to occur. The CoPC screening cannot, however, establish that an unacceptable risk exists at the site. Rather, it identifies which chemicals, if any, require a more site-specific analysis to determine if risks are elevated.

The remainder of this section describes the human health CoPC screening and the selection of CoPCs for each of the environments being evaluated (i.e., the terrestrial, stream, and marine environments).

3.3.1 Terrestrial Environment

The CSM describes the exposure pathways relevant for assessing potential risks to human health in the terrestrial environment. As indicated in the CSM, the primary environmental media to which people could be exposed in the terrestrial environment are soil and dust. This includes soil on or near the road and port industrial areas, re-suspended dust in the air, and dust on plant and animal surfaces. There is little bare soil in the tundra outside of the road and port, and people would come into relatively little contact with soil underneath the tundra mat. In addition, chemical concentrations in soil away from the road and port would be lower than on the road and port industrial area if those chemical concentrations are influenced by fugitive dust deposition. A conservative screening, therefore, includes soil samples from the port, road, and road shoulder. These data are summarized in Table 3-4. The remainder of this section summarizes the comparison of site soil data with chemical concentrations in soil not impacted by the DMTS, as well as the comparison to health-protective risk-based screening levels.

3.3.1.1 Comparison of Site Soil Data with Reference Data

Soil samples were collected from excavation sites used to supply material for road repair. Because these areas are not believed to be impacted by fugitive dust or other mine activities, the chemical concentrations from these locations are considered representative of pre-mine or reference conditions. Thus, site soil chemical concentrations were compared to these reference data to determine which constituents are present at the site above pre-mine conditions. The results of this comparison, as summarized in Table 3-4, indicate that 11 constituents (barium, cadmium, calcium, fluoride, lead, manganese, mercury, silver, strontium, thallium, and zinc) are statistically elevated compared to reference concentrations.

3.3.1.2 Comparison of Site Data with Risk-Based Screening Values

Maximum surface soil concentrations from the road and port were also compared with residential screening levels, as prescribed in DEC (2000). DEC (2000) indicates that site concentrations should be screened against residential screening levels, which are derived by dividing the cleanup levels provided in Table B1 of DEC (2003) by an additional safety factor

of 10. This safety factor corresponds to DEC's requirement that screening levels, unlike cleanup levels, be based on a target risk of 1×10^{-6} for carcinogens and a target hazard quotient (THQ) of 0.1 for non-carcinogens. These screening levels were derived assuming that a person would be living at the site and that all incidental soil ingestion from birth to 30 years of age would occur at the site. Furthermore, DEC (2000) indicates that risk-based screening levels should be calculated for site target chemicals for which there is no cleanup level listed in Table B1 of DEC (2003) using the residential cleanup level formula and assumptions provided in DEC (2002), but with a target risk of 1×10^{-6} or a target hazard quotient of 0.1. For chemicals that cause cancer, the residential risk-based screening level is calculated using the following formula:

$$\text{Residential Screening Level (cancer, mg/kg)} = \frac{\text{TR} \times \text{AT}}{\text{CSF} \times 10^{-6}\,(\text{mg/kg}) \times \text{EF} \times \text{IF}}$$

where:

TR	=	target cancer level (unitless)	=	10^{-6}
AT	=	averaging time (days)	=	25,550
CSF	=	cancer slope factor $(\text{mg/kg-day})^{-1}$	=	chemical specific
EF	=	exposure frequency (days/year)	=	200 for arctic zone
IF	=	age-adjusted ingestion factor (mg-year/kg-day)	=	114

The age-adjusted soil ingestion factor adjusts soil ingestion to take into account different soil ingestion rates and body weights for children and adults, and is calculated as follows:

$$\text{IF} = \frac{\text{IR}_{1-6} \times \text{ED}_{1-6}}{\text{BW}_{1-6}} + \frac{\text{IR}_{7-31} \times \text{ED}_{7-31}}{\text{BW}_{7-31}}$$

where:

$\text{IR}_{1\text{-}6}$	=	soil ingestion rate, ages 1–6 (mg/day)	=	200
$\text{ED}_{1\text{-}6}$	=	exposure duration, ages 1–6 (years)	=	6
$\text{BW}_{1\text{-}6}$	=	body weight, ages 1–6 (kg)	=	15
$\text{IR}_{7\text{-}31}$	=	soil ingestion rate, ages 7–31 (mg/day)	=	100
$\text{ED}_{7\text{-}31}$	=	exposure duration, ages 7–31 (years)	=	24
$\text{BW}_{7\text{-}31}$	=	body weight, ages 7–31 (kg)	=	70

For chemicals with health effects other than cancer, the residential risk-based screening level is calculated using the following formula:

$$\text{Residential Screening Level (non-cancer, mg/kg)} = \frac{\text{THQ} \times \text{BW} \times \text{AT} \times \text{RfD}}{10^{-6}\,(\text{mg/kg}) \times \text{EF} \times \text{ED} \times \text{IR}}$$

where:

THQ	=	target hazard quotient (unitless)	=	0.1
BW	=	body weight, child (kg)	=	15
AT	=	averaging time (days)	=	2,190
RfD	=	reference dose (mg/kg-day)	=	chemical specific
EF	=	exposure frequency (days/year)	=	200 for arctic zone
ED	=	exposure duration (years)	=	6
IR	=	ingestion rate, child (mg/day)	=	200

The chemical-specific cancer slope factors (CSFs) and reference doses (RfDs) are provided in DEC (2002). The cleanup level for lead listed in Table B1 of DEC (2003) was not calculated using this methodology, but rather is the product of modeling using EPA's integrated exposure uptake/biokinetic (IEUBK) child lead model. The IEUBK guidance (U.S. EPA 1996c) calls for central tendency (i.e., average) inputs and the model has been validated using central tendency input parameters. The screening level represents a soil concentration that corresponds to a distribution of blood lead levels with an upper end (i.e., 95th percentile) at the target blood lead level of 10 μg/dL. Because the lead cleanup level was not derived using an RfD and THQ, use of the additional safety factor would be inconsistent with the purpose and application of the IEUBK model. Therefore, the screening level for lead is equivalent to the cleanup level. Human health screening levels for soil for all site target analytes are presented in Table 3-14. Maximum site soil concentrations of 10 chemicals exceeded risk-based screening levels (aluminum, antimony, arsenic, barium, cadmium, iron, lead, manganese, thallium, and zinc).

Although DEC (2003) guidance requires screening all sites using residential screening assumptions, there are no residences near the site and residential use is not expected in the future. In order to evaluate site concentrations from the perspective of exposures that are more likely to occur at the site, maximum site chemical concentrations were also compared to health-based screening levels assuming non-residential exposure. Specifically, risk-based screening levels were calculated using the industrial cleanup level formula and assumptions provided in DEC (2002), but with a target risk of 1×10^{-6} or a target hazard quotient of 0.1. Because the non-residential screening values incorporate a high degree of exposure (i.e., 200 days per year for the arctic zone, for 25 years) these screening levels would still be a conservative means to evaluate the lower frequency exposures that might occur at the site. This comparison was not used to screen out chemicals from the site, but rather to provide a frame of reference for evaluating the site under more realistic, yet still conservative, conditions.

For chemicals that cause cancer, the non-residential risk-based screening level is calculated using the following formula:

$$\text{Non}-\text{Residential Screening Level (cancer, mg/kg)} = \frac{\text{TR}\times\text{BW}\times\text{AT}}{\text{CSF}\times10^{-6}(\text{mg/kg})\times\text{EF}\times\text{ED}\times\text{IR}}$$

where:

TR	=	target cancer level (unitless)	=	10^{-6}
BW	=	body weight (kg)	=	70
AT	=	averaging time (days)	=	25,550
CSF	=	cancer slope factor $(mg/kg\text{-}day)^{-1}$	=	chemical specific
EF	=	exposure frequency (days/year)	=	200
ED	=	exposure duration (years)	=	30
IR	=	soil ingestion rate, adult (mg/day)	=	50

For chemicals with health effects other than cancer, the residential risk-based screening level is calculated using the following formula:

$$\text{Non}-\text{Residential Screening Level}\,(\text{non}-\text{cancer, mg/kg}) = \frac{\text{THQ} \times \text{BW} \times \text{AT} \times \text{RfD}}{10^{-6}\,(\text{mg/kg}) \times \text{EF} \times \text{ED} \times \text{IR}}$$

where:

THQ	=	target hazard quotient (unitless)	=	0.1
BW	=	body weight, adult (kg)	=	70
AT	=	averaging time (days)	=	9,125
RfD	=	reference dose (mg/kg-day)	=	chemical specific
EF	=	exposure frequency (days/year)	=	200 for arctic zone
ED	=	exposure duration (years)	=	25
IR	=	soil ingestion rate, adult (mg/day)	=	50

Arsenic, cadmium, and lead were present at concentrations exceeding non-residential risk-based screening levels. Arsenic exceeded the non-residential screening level in 1 out of 75 samples, cadmium in 2 out of 236 samples, and lead in 168 out of 479 samples. With the exception of one lead sample near the ambient air boundary of the mine, all exceedances of non-residential screening levels occurred within the ambient air boundary of the port.

3.3.1.3 Selection of Human Health CoPCs for the Terrestrial Environment

Maximum site soil concentrations of six chemicals (antimony, barium, cadmium, lead, thallium, zinc) exceeded both their risk-based screening level and the reference concentrations (in the case of antimony, there were too few detections in site samples to statistically compare with reference samples) (Table 3-14). Thallium exceeded the screening level in only one of 12 samples, and by less than 2-fold (maximum concentration of 1.32 mg/kg vs. screening value of 0.9 mg/kg). In addition, the single thallium exceedance occurred within the ambient air boundary of the mine. Antimony exceeded the screening level in only one of 40 samples, and by less than 3-fold (maximum concentration of 14.8 mg/kg vs. screening value of 5.5 mg/kg).

The single antimony exceedance occurred near CSB2 at the port site. Given the low frequencies of exceedance of screening levels, the small magnitude of the exceedances, the location of the exceedances (within the mine solid waste permit boundary for thallium and at CSB2 for antimony), the conservative nature of the screening levels (i.e., assuming residential exposure), and the additional 10-fold safety factor applied, the levels of antimony and thallium present at the site are highly unlikely to pose a human health risk at the site. Thus, antimony and thallium will not be retained as human health CoPCs for the terrestrial environment. Barium, cadmium, lead, and zinc will be retained as human health CoPCs for the terrestrial environment.

Sample screening for the four terrestrial environment CoPCs is depicted spatially in Figures 3-5 through 3-8. For cadmium (Figure 3-6), lead (Figure 3-7), and zinc (Figure 3-8), only one or two samples exceeding the residential screening criteria were located outside the port facilities area or the mine solid waste permit boundary. For barium (Figure 3-5), five of six samples located outside the ambient air boundary of the port exceeded the residential screening criterion, but none exceeded the non-residential criterion (Figure 3-5).

3.3.2 Freshwater Environment

As described in the CSM, the primary environmental medium of concern in the freshwater environment is surface water in streams in the vicinity of the road and port. Streams near the road drain into the Wulik River, which is the drinking water source for Kivalina. Because the risk assessment is designed to evaluate potential impacts of fugitive dust from the DMTS, this assessment has focused on surface water nearest to the road, even though any potential fugitive dust-associated chemical concentrations in streams near the DMTS would be diluted greatly when mixing with the Wulik River. A person could potentially drink water directly from a stream near the DMTS while engaged in subsistence activities. However, the criteria that are used for the CoPC screening in the freshwater environment assume that all of a person's drinking water would come from the water body being evaluated, which would not be the case for streams near the DMTS. Thus, chemical concentrations from streams in the vicinity of the DMTS were used to screen CoPCs in the freshwater environment. In addition, fish in these streams and from the Wulik River provide a subsistence food source for people living in the area. Thus, stream chemical concentrations were also compared to ambient water quality criteria (AWQC) protective of drinking water and bioaccumulation into fish, when AWQC were available. This section describes the results of that comparison, as well as a comparison to chemical concentrations in stream surface water from a reference area not affected by the DMTS.

3.3.2.1 Comparison of Site Stream Water Data with Reference Data

Water samples were collected from the terrestrial reference area to the south (upwind) of the DMTS road (see Figure 3-4). Unfiltered site stream surface water chemical concentrations were compared to reference stream surface water data to determine which constituents were present at the site above pre-mine conditions. The results of this comparison, as summarized in Table 3-7, indicate that fluoride and molybdenum are statistically elevated compared to reference concentrations. Arsenic, chromium, mercury, and silver were not detected in any site stream surface water samples. Statistical comparisons to reference samples could not be made for

antimony, cadmium, lead, selenium, thallium, tin, vanadium, and zinc because there were too few detected samples. However, the maximum site vanadium concentration is lower than the maximum reference concentration (Table 3-15), indicating that site vanadium concentrations are consistent with reference conditions.

3.3.2.2 Comparison of Site Stream Water Data with Risk-Based Screening Values

Maximum site surface water concentrations from streams in the vicinity of the DMTS were compared with residential screening levels, as prescribed in DEC (2000). DEC (2000) indicates that site concentrations should be screened against residential screening levels, which are derived by dividing the cleanup levels provided in Table B1 of DEC (2003) by an additional safety factor of 10. This safety factor corresponds to DEC's requirement that screening levels, unlike cleanup levels, be based on a target risk of 1×10^{-6} for carcinogens and a THQ of 0.1 for non-carcinogens. These screening levels were derived assuming use of the water body as the primary drinking water source in a residential setting. Furthermore, DEC (2000) indicates that risk-based screening levels should be calculated for site target chemicals for which there is no cleanup level listed in DEC (2003), using the residential cleanup level formula and assumptions provided in DEC (2002), but with a target risk of 1×10^{-6} or a target hazard quotient of 0.1. For chemicals that cause cancer, the residential risk-based screening level is calculated using the following formula:

$$\text{Residential Screening Level (cancer, mg/kg)} = \frac{\text{TR} \times \text{BW} \times \text{AT}}{\text{CSF} \times \text{IR} \times \text{EF} \times \text{ED}}$$

where:

TR	=	target cancer level (unitless)	=	10^{-6}
BW	=	body weight (kg)	=	70
AT	=	averaging time (days)	=	25,550
CSF	=	cancer slope factor $(\text{mg/kg-day})^{-1}$	=	chemical-specific
IR	=	water ingestion rate (liters/day)	=	2
EF	=	exposure frequency (days/year)	=	350
ED	=	exposure duration (years)	=	30

For chemicals with health effects other than cancer, the residential risk-based screening level is calculated using the following formula:

$$\text{Residental Screening Level (non-cancer, mg/kg)} = \frac{\text{THQ} \times \text{RfD} \times \text{BW} \times \text{AT}}{\text{IR} \times \text{EF} \times \text{ED}}$$

where:

THQ	=	target hazard quotient (unitless)	=	0.1
AT	=	averaging time (days)	=	10,950
RfD	=	reference dose (mg/kg-day)	=	chemical-specific

BW, IR, EF, and ED are as described above.

The chemical-specific CSFs and RfDs are provided in DEC (2002). Human health screening levels for surface water for all site target analytes are presented in Table 3-15. Maximum site stream water concentrations of aluminum, barium, iron, lead, and thallium exceeded residential drinking water risk-based screening levels. However, the frequency of exceedance for all five of these chemicals was low (<5 percent for aluminum, iron, and lead; <8 percent for barium and thallium).

Stream surface water chemical concentrations were also compared to AWQC protective of human consumption of both water and fish from the water body (Table 3-16). AWQC were available for seven chemicals. Where no AWQC were available, chemical concentrations were compared to screening levels developed by the Washington State Department of Ecology (WDOE) protective of bioaccumulation into, and human consumption of fish from, the water body (WDOE 1996). WDOE criteria were available for an additional three analytes. Both the AWQC and the WDOE criteria were divided by a safety factor of 10 to be consistent with DEC screening guidelines. In all cases where AWQC or WDOE criteria were available (i.e., antimony, arsenic, cadmium, chromium, copper, nickel, selenium, silver, thallium, and zinc), the site maximum detected chemical concentration was below those criteria.

3.3.2.3 Selection of Human Health CoPCs for the Freshwater Environment

Only one chemical, thallium, had a maximum site stream surface water concentration that both exceeded its risk-based screening level and could not be determined to be consistent with reference conditions. Given the low frequency of exceedance of the screening level (i.e., 2 of 27), the small magnitude of exceedance (0.55 μg/L vs. 0.2 μg/L), the fact that chemical concentrations in streams near the road would be greatly diluted when joining the Wulik River, the conservative nature of the screening levels (i.e., assuming residential drinking water exposure), and the additional 10-fold safety factor applied, the levels of thallium present in site surface water are highly unlikely to pose a human health risk at the site. Thus, there are no human health CoPCs associated with water consumption in the freshwater environment.

Screening criteria protective for fish consumption were available for 10 chemicals. In all cases, maximum site stream surface water concentrations were below those criteria. Only four chemicals that did not have fish consumption criteria (fluoride, lead, molybdenum, and tin) also could not be screened out by comparison to reference conditions (Table 3-16). In all cases (with the exception of arsenic), the available fish consumption screening criteria are greater than the drinking water screening levels. Given that none of these four chemicals would be expected to bioaccumulate significantly in fish, screening criteria based on fish consumption would also be expected to be greater than drinking water screening levels if bioconcentration factors were available for the four chemicals to calculate them. Therefore, fluoride, lead, molybdenum, and

tin will not be retained as CoPCs and there are, thus no human health CoPCs associated with fish consumption in the freshwater environment.

3.3.3 Coastal Lagoon and Marine Environments

As described in the CSM, the primary potential human exposure pathway in the marine environment would be bioaccumulation of chemicals in the food chain, and subsequent consumption of marine animals by people. Thus, chemical concentrations in lagoon and marine water near the port were compared to water quality criteria protective of human consumption of seafood. This section describes the results of that comparison, as well as a comparison to chemical concentrations in lagoon and marine water from areas not impacted by the DMTS.

3.3.3.1 Comparison of Site Lagoon and Marine Data with Reference Data

As described in Section 3.2.6.2, lagoon water samples were collected from the Control Lagoon and Reference Lagoon, located approximately 2 miles and 5 miles, respectively, to the southeast (in the prevailing upwind direction) of the port site facilities. As described in Section 3.2.7.2, marine water samples were collected from the marine reference area located approximately 3 miles to the south of the port site (in the prevailing upwind and upcurrent direction). Site lagoon and marine water chemical concentrations were compared to reference data to determine which constituents are present at the site above pre-mine conditions.

The results of the lagoon water reference comparison, as summarized in Table 3-11, indicate that antimony, fluoride, lead, and molybdenum are statistically elevated compared to reference conditions. Mercury was not detected in any site or reference sample. A statistical comparison to reference samples could not be made for tin because it was detected in only one of eight samples in site samples and was not detected in any reference sample.

The results of the marine water reference comparison, as summarized in Table 3-13, indicate that only selenium is statistically elevated compared to reference concentrations. Chromium, mercury, nickel, and zinc were not detected in any site sample. Statistical comparisons to reference samples could not be made for copper, thallium, tin, and vanadium because there were too few detected samples in site and reference data. However, the maximum site tin and vanadium concentrations were lower than the maximum reference concentrations, indicating that site tin and vanadium concentrations are consistent with reference conditions. Thus, selenium, copper, and thallium cannot be screened out by comparison with reference conditions.

3.3.3.2 Comparison of Site Lagoon and Marine Data with Risk-Based Screening Values

Maximum site lagoon and marine water data were compared to AWQC protective of bioaccumulation in, and consumption of, seafood (U.S. EPA 2002c). The AWQC were modified, when necessary, to include a THQ of 0.1 or a target risk of 10^{-6}, to be consistent with DEC (2000) guidance for screening levels. Fish consumption AWQC are available only for antimony, arsenic, nickel, selenium, thallium, and zinc. In addition, site concentrations were compared to surface water criteria published by WDOE (1996). The WDOE surface water

criteria are based on bioaccumulation into, and human consumption of, seafood. WDOE criteria are available for arsenic, cadmium, chromium, copper, nickel, silver, thallium, and zinc. In both lagoon water and marine water, only arsenic exceeded its AWQC or WDOE surface water criterion (Tables 3-17 and 3-18, respectively).

As described in the CSM, people would come into very little direct contact with lagoon sediments and not at all with marine sediments at the site. Thus, it would not be appropriate to use soil ingestion screening values to screen lagoon and marine sediments, even if they were modified to assume a lower sediment ingestion rate. The primary potential exposure pathway in the lagoons and marine environment at the site would be bioaccumulation of chemicals in the food chain, and consumption of marine biota. There are no screening values available that address this pathway. The SQS developed for the Washington State Sediment Management Standards (WDOE 1995), though based on protection of benthic infauna, are commonly applied to marine sediments and assumed to also be protective of human health. Washington State regulations, in fact, explicitly state that the SQS are protective of human health (WDOE 1995). As described in detail in the ecological CoPC screening section (see Section 3.5), all chemicals except zinc in the lagoons and all chemicals in the marine environment are screened out when sediment chemical concentrations are compared to SQS. The maximum zinc concentration in lagoon sediments (1,590 mg/kg), however, is still lower than the soil screening criteria for zinc of 4,100 mg/kg. Thus, even with the intense direct contact assumed in the soil screening criteria, human exposure to the zinc concentrations in lagoon sediments would not pose a risk to human health.

3.3.3.3 Selection of Human Health CoPCs for the Lagoon and Marine Environments

There were no chemicals in lagoon or marine water with a maximum site concentration that exceeded both the reference concentrations and their risk-based screening levels. Thus, there are no lagoon or marine water CoPCs. This result, along with the lack of lagoon or marine sediment CoPCs when screened using Washington State SQS (see Sections 3.5.4 and 3.5.5), indicates that fugitive dust from the DMTS has not significantly impacted the lagoon or marine environment near the site. Therefore, the lagoon and marine environment will not be further evaluated in the HHRA.

3.4 Selection of Human Health CoPCs

In the preceding section, site environmental media were screened against reference concentrations and conservative, health-based screening levels for the constituents present in the source material (i.e., the chemicals in the lead and zinc concentrates transported along the DMTS). The following chemicals were retained as CoPCs:

- Terrestrial environment: barium, cadmium, lead, zinc
- Freshwater environment: no CoPCs
- Marine environment: no CoPCs.

3.5 Ecological Screening Assessment

Two screening approaches were used to identify CoPCs for ecological receptors. The maximum concentrations of chemicals in tundra soil, sediment, and surface water in different environments at the site were compared against multiple ecological screening benchmarks. Screening benchmarks represent ambient concentrations of a chemical that, if exceeded, could indicate the potential for-adverse effects to lower trophic-level ecological receptors such as plants and invertebrates. In addition, screening-level food web models were developed to estimate dietary exposures to chemicals for representative avian and mammalian receptors that may feed at the site. Food web models were developed for tundra vole, representing terrestrial herbivores; red-throated loon, representing avian piscivores; river otter, representing mammalian piscivores; common snipe, representing freshwater avian invertivores; and black-bellied plover, representing marine avian invertivores. Daily chemical exposures for each receptor were compared to no-effect based TRVs to evaluate whether exposures to maximum chemical concentrations in tundra soil, stream sediment, and food could potentially result in adverse ecological effects.

The screening assessment does not result in a quantitative risk characterization. Only the absence (not the presence) of risk can be established by a screening assessment alone. If the possibility of adverse effects cannot be ruled out using screening approaches, further assessment may be required for those exposure pathways and receptor communities. The following sections describe the screening results for the media represented in each environment and the results of the wildlife exposure models.

3.5.1 Terrestrial Tundra Environment

Tundra soil data were compared to Oak Ridge National Laboratory (ORNL) toxicological benchmarks for effects on terrestrial plants (Efroymson et al. 1997a) and earthworms and microbial heterotrophs (Efroymson et al. 1997b). The ORNL screening benchmarks approximate the 10th percentile of lowest-observed-effect concentrations reported in studies that examined the effects of chemicals on vascular plant growth or production (yield) (Efroymson et al. 1997a), earthworm survival, growth, and reproduction (Efroymson et al. 1997b), or soil microflora community functioning, including carbon mineralization, nitrogen transformation, and enzyme activities (Efroymson et al. 1997b). Soil screening benchmarks are presented in Table 3-19. Benchmarks for toxicological effects in terrestrial plants have not been developed for iron or strontium in soil. Benchmarks for toxicological effects on earthworms have not been developed for aluminum, antimony, barium, cobalt, iron, manganese, molybdenum, silver, strontium, thallium, tin, or vanadium. Benchmarks for toxicological effects on microbial heterotrophs have not been developed for antimony, strontium, or thallium. Plant and microbial benchmarks for fluorine were used to screen fluoride data from the site. Tundra soil screening results are summarized in Table 3-19. Maximum concentrations of all chemicals for which there are ORNL phytotoxicity benchmarks exceeded their benchmarks, with the exception of copper, fluoride, and tin. Maximum concentrations of seven chemicals (arsenic, cadmium, chromium, copper, lead, mercury, and zinc) exceeded the ORNL earthworm benchmarks. Maximum nickel and selenium concentrations in tundra soil were below ORNL earthworm benchmarks. Maximum concentrations of 10 chemicals (aluminum, arsenic, barium, cadmium,

chromium, iron, lead, manganese, vanadium, and zinc) exceeded the ORNL benchmarks for microbial heterotrophs, while maximum concentrations of nine chemicals (cobalt, copper, fluoride, mercury, molybdenum, nickel, selenium, silver, and tin) were below the benchmarks.

For several chemicals, exceedances of screening benchmarks occurred predominantly in tundra soil samples collected near the port facility. Antimony, cobalt, copper, and silver concentrations exceeded screening benchmarks at the port site only and did not exceed benchmarks in samples collected along DMTS road transects outside the port area. Five out of six exceedances of the ORNL terrestrial plant benchmarks for arsenic and nickel occurred at the port site; the remaining exceedances occurred at transect station TT4-0010 within the solid waste permit boundary at the mine (Figure 3-2). In contrast, molybdenum exceeded the ORNL phytotoxicity benchmark in four samples, three of which were collected at stations along transect TT4 but only one of which was collected at the port. Chemicals such as cadmium, lead, and zinc exceeded screening benchmarks at many terrestrial transect stations and port site stations (Figures 3-9, 3-10, and 3-11).

3.5.2 Streams

3.5.2.1 Stream Sediment

Streambed surface sediment data were compared to freshwater threshold effect concentrations (TECs) and probable effect concentrations (PECs) developed by MacDonald et al. (2000). The TEC is the sediment concentration below which adverse effects to benthic organisms are not expected; the PEC is the sediment concentration above which adverse effects to benthic organisms are expected to occur frequently, according to MacDonald et al. (2000). Sediment concentrations were also compared to no-effect concentrations (NECs) derived by Ingersoll et al. (1996) from 28-day toxicity tests on the amphipod *Hyalella azteca*. The NEC is the sediment concentration of a given chemical above which a statistically significant effect is always observed (Ingersoll et al. 1996). Freshwater sediment screening benchmarks are presented in Table 3-20. Benchmarks are not available for a number of chemicals, including antimony, barium, cobalt, fluoride, molybdenum, selenium, silver, strontium, thallium, tin, and vanadium. No TEC or PEC screening value is available for aluminum (MacDonald et al. 2000), and no NEC value is available for mercury (Ingersoll et al. 1996).

Table 3-20 summarizes the results of the stream sediment screening. Maximum concentrations of five chemicals (arsenic, cadmium, lead, nickel, and zinc) exceeded their TECs. Maximum lead and nickel concentrations also exceeded the PEC and NEC. Maximum concentrations of six chemicals (aluminum, chromium, copper, iron, manganese, and mercury) in stream sediment did not exceed any screening benchmarks. While nickel and zinc concentrations exceeded their TECs at one or more stations in each creek sampled (zinc results shown in Figure 3-11), arsenic, cadmium, and lead concentrations only exceeded their TECs in sediment collected from Anxiety Ridge Creek (cadmium and lead results shown in Figures 3-9 and 3-10). Lead exceeded its NEC in one sample collected in Anxiety Ridge Creek upstream of the DMTS road (Figure 3-10).

3.5.2.2 Stream Surface Water

Chemical concentrations in unfiltered stream water were compared to EPA's national AWQC criterion continuous concentration (CCC) and criteria maximum concentration (CMC) values for the protection of freshwater aquatic life, such as aquatic invertebrates and fish (U.S. EPA 2002c). The CCC is the highest water concentration of a given chemical to which an aquatic community can be exposed indefinitely without adverse effect; the CMC is the highest water concentration of a given chemical to which an aquatic community can be exposed briefly without adverse effect (U.S. EPA 2002c). The AWQC for cadmium, chromium, copper, lead, nickel, silver, and zinc are hardness-dependent and were adjusted in the screening to reflect site-specific water hardness. Table 3-21 presents freshwater AWQC corresponding to a default hardness of 100 mg/L as calcium carbonate ($CaCO_3$) and reported on a total recoverable basis (U.S. EPA 2002c). The AWQC for chromium(VI) were conservatively used to screen total chromium data from the site. There are no AWQC for antimony, barium, cobalt, fluoride, manganese, molybdenum, strontium, thallium, tin, or vanadium.

Results of the stream water screening are summarized in Table 3-21. Maximum concentrations of five chemicals (aluminum, cadmium, iron, lead, and zinc) exceeded the CCC; aluminum and zinc concentrations also exceeded their respective CMCs at one or more stations. Chemical concentrations exceeded benchmarks in various creeks with the exception of zinc, which exceeded the CCC at only one station located downstream of the DMTS road in Tutak Creek (Figure 3-4). Maximum detected concentrations of three chemicals (copper, nickel, and selenium) did not exceed screening benchmarks. Arsenic, chromium, mercury, and silver were undetected in all samples, and values equal to half the detection limit did not exceed screening benchmarks.

3.5.3 Tundra Ponds

3.5.3.1 Tundra Pond Sediment

Chemical concentrations in tundra pond surface sediment were compared to the TEC, PEC, and NEC (MacDonald et al. 2000; Ingersoll et al. 1996). Results of the tundra pond sediment screening are summarized in Table 3-22. Maximum concentrations of six chemicals (cadmium, copper, lead, mercury, nickel, and zinc) exceeded the TEC. Cadmium, lead, mercury, and zinc concentrations also exceeded the PEC. Maximum concentrations of four chemicals (cadmium, lead, nickel, and zinc) exceeded the NEC. Arsenic, chromium, iron, and manganese concentrations in tundra pond sediment did not exceed any toxicity thresholds.

Zinc concentrations in all tundra pond sediments sampled exceeded the TEC (Figure 3-11). Cadmium, copper, lead, and mercury concentrations exceeded benchmarks in the two tundra ponds located approximately 100 m from the DMTS road but not in the two ponds located approximately 1,000 m from the road (cadmium and lead results shown in Figures 3-9 and 3-10). Copper and mercury exceedances in sediment occurred only at station TP1-0100 near the port facility (Figure 3-1; Photograph 4). For all chemicals, exceedances of the NEC occurred only at station TP1-0100.

3.5.3.2 Tundra Pond Surface Water

Chemical concentrations in unfiltered tundra pond water were compared to the freshwater CCC and CMC values (U.S. EPA 2002c), as summarized in Table 3-23. Maximum concentrations of six chemicals (aluminum, cadmium, copper, iron, lead, and zinc) exceeded the CCC, and the maximum zinc concentration also exceeded its CMC value. Maximum concentrations of arsenic, chromium, and nickel did not exceed AWQC, and mercury, selenium, and silver were undetected in all samples. Cadmium and zinc concentrations exceeded screening benchmarks at station TP1-1000 only (station location shown in Figure 3-4), while exceedances for lead were more widespread.

3.5.4 Coastal Lagoons

3.5.4.1 Lagoon Sediment

Chemical concentrations in coastal lagoon surface sediment were compared to effects range-low (ERL) and effects range-median (ERM) guideline values developed by Long et al. (1995) for marine sediment and the Washington State marine SQS (WAC 173–204). The ERL represents the 10th percentile of the distribution of effects data assembled from studies examining endpoints ranging from hepatic lesions to mortality; the ERM represents the 50th percentile of the effects data distribution. The ERL is intended to be the sediment concentration of a given chemical below which adverse effects to marine life rarely occur, while the ERM is intended to be the sediment concentration equal to or above which adverse effects to marine life frequently occur (Long et al. 1995). Washington State SQS are no-effects levels, or levels at or below which sediments have no adverse effects on biological resources (WAC 173–204). They are sediment quality goals for the State of Washington, but have also been applied at sites in Alaska (Exponent 1999). Lagoon sediment screening benchmarks are presented in Table 3-24.

The results of the lagoon sediment screening are summarized in Table 3-24. Maximum concentrations of five chemicals (arsenic, cadmium, lead, nickel, and zinc) exceeded their ERL values, and maximum lead and zinc concentrations also exceeded their ERM values. Maximum cadmium and zinc concentrations exceeded their SQS values. Maximum concentrations of four chemicals (chromium, copper, mercury, and silver) in lagoon sediment did not exceed any screening benchmarks.

Spatial patterns and frequencies of exceedance varied by chemical. Cadmium, lead, and zinc concentrations in sediment exceeded their ERL values in multiple lagoons (Figures 3-9, 3-10, and 3-11). Only the maximum cadmium concentration, measured in Port Lagoon North, exceeded its SQS (Figure 3-9; Photograph 3), while zinc exceedances occurred at four stations located in three lagoons (Port Lagoon North, Port Lagoon South, and the North Lagoon; Figure 3-11). Arsenic exceedances were limited to the Ipiavik Lagoon and the North Lagoon, and nickel exceedances were found only in the North Lagoon (Figure 3-3 shows station and lagoon locations).

3.5.4.2 Lagoon Surface Water

Chemical concentrations in unfiltered lagoon water were compared to the saltwater CCC and CMC values (U.S. EPA 2002c). Results of the lagoon surface water screening are summarized in Table 3-25. Maximum arsenic and zinc concentrations exceeded the CCC and the CMC values, and the maximum nickel concentration exceeded the CCC. Maximum concentrations of six chemicals (cadmium, chromium, copper, lead, selenium, and silver) did not exceed saltwater AWQC. Mercury was undetected in all lagoon water samples, and values reported at half the detection limit did not exceed screening benchmarks.

The maximum zinc concentration, measured in water collected at one station in the North Lagoon, was the only zinc value that exceeded screening benchmarks (Figure 3-11). The spatial patterns of arsenic and nickel exceedances were similar to the results for lagoon sediment; all arsenic and nickel exceedances occurred in Ipiavik Lagoon sediment (Figure 3-3 shows Ipiavik Lagoon station locations).

3.5.5 Marine Environment

3.5.5.1 Marine Sediment

Surface sediment data from nearshore and offshore areas around the port facility were compared with the ERL, ERM, and SQS. Table 3-26 summarizes the results of the marine sediment screening. Maximum concentrations of eight chemicals (arsenic, cadmium, copper, lead, mercury, nickel, silver, and zinc) exceeded the ERL, and maximum cadmium, lead, and zinc concentrations also exceeded the ERM. Maximum concentrations of four chemicals (cadmium, lead, mercury, and zinc) exceeded their SQS. Chromium concentrations in marine sediment did not exceed any screening benchmarks.

Copper, mercury, and silver concentrations exceeded the ERL at one station located directly below the shiploader (Figure 3-3), while exceedances for chemicals such as arsenic, cadmium, and nickel were more widespread but interspersed with stations where the ERL was not exceeded (cadmium results shown in Figure 3-9). Exceedances of the SQS were localized to stations around the shiploader. Cadmium, lead, and mercury exceeded their SQS at one station located directly below the shiploader, and zinc exceeded its SQS at three stations surrounding the shiploader (cadmium, lead, and zinc results shown in Figures 3-9 through 3-11).

3.5.5.2 Marine Surface Water

Chemical concentrations in unfiltered marine water were compared to the saltwater CCC and CMC values (U.S. EPA 2002c), as summarized in Table 3-27. The maximum copper concentration, measured at a station located directly below the shiploader (Figure 3-4), exceeded its CCC and CMC values. Maximum concentrations of arsenic, cadmium, lead, selenium, and silver were below the CCC and CMC values. Chromium, mercury, nickel, and zinc were undetected in all marine water samples. A value equivalent to half of the maximum detection limit for nickel exceeded the CCC.

3.5.6 Wildlife

Identification of CoPCs for higher trophic-level wildlife (birds and mammals) is accomplished by using available site data in screening-level food web models to evaluate the exposure potential for representative terrestrial and aquatic receptors. Conservative assumptions, as described below, are made throughout this modeling exercise to preclude the possibility of a false negative finding at the screening stage. Preliminary evaluation of the exposure potential for avian and mammalian receptors will be accomplished using simple deterministic food-web exposure models consistent with EPA's wildlife exposure guidance (U.S. EPA 1993; 61 Fed. Reg. 47552). The food-web model estimates dietary exposure as a body-weight-normalized total daily dose for each receptor species. The general structure of the food-web exposure model is described by the following equation:

$$IR_{chemical} = \frac{\sum_i (C_i \times M_i \times A_i \times F_i)}{W}$$

where:

$IR_{chemical}$ = total ingestion rate of chemical from all dietary components (mg dry weight/kg body weight/day)

C_i = concentration of the chemical in a given dietary component or inert medium (mg/kg dry weight)

M_i = rate of ingestion of dietary component or inert medium (kg dry weight/day)

A_i = relative gastrointestinal absorption efficiency for the chemical in a given dietary component or inert medium (fraction)

F_i = fraction of the daily intake of a given dietary component or inert medium derived from the site (unitless area-use factor)

W = body weight of receptor species (kg).

The term $IR_{chemical}$ can be expanded to specify each ingestion medium, which includes one or more primary food items, drinking water, and incidentally ingested sediment or soil:

$$IR_{chemical} = [\Sigma (C_{food} \times M_{food} \times A_{food} \times F_{food}) + (C_{water} \times M_{water} \times A_{water} \times F_{water}) + (C_{sediment/soil} \times M_{sediment/soil} \times A_{sediment/soil} \times F_{sediment/soil})]/W$$

The model provides an estimated total dietary exposure to chemicals resulting from consumption of food and the incidental ingestion of soil or sediment on a mg chemical/kg body-weight-day basis. For all the receptors modeled, the screening-level exposure calculation assumes that the entire diet comes from the study area ($F_i = 1$), and that 100 percent of the chemical ingested in food is absorbed ($A_i = 1$). The maximum chemical concentrations reported in food items or environmental media are used in the exposure estimates (data tables are included in Appendix C). These conservative assumptions represent a worst-case exposure scenario, thus using these values results in protective exposure estimates that are appropriate for a screening-level assessment. Water ingestion was not included in the exposure analysis, but

because chemical concentrations in water are low, exposure via water is minimal compared to exposure via food and soil/sediment ingestion, and results are not affected by omission of this pathway.

For all representative receptors, exposure estimates are compared to no-observed-adverse-effects level (NOAEL) TRVs to calculate hazard quotients. For screening purposes, if the ratio of exposure to the TRV is less than 1.0, then the chemical is not considered likely to cause adverse effects to upper trophic-level receptors, and will not be retained as a CoPC. Chemicals where the hazard quotient exceeds 1.0 in these conservative food web models will be retained as CoPCs in the baseline risk assessment. The TRVs used in the screening models are presented in Table 3-28.

3.5.6.1 Terrestrial Wildlife

To calculate point estimates of dietary exposure it is necessary to select representative receptors. The only terrestrial food items that have been analyzed for CoPCs are several plant species (moss, lichen, willow, berries); therefore, data are only available to directly evaluate exposure to herbivorous receptors. However, due to the elevated chemical concentrations in plants, particularly moss, exposure of herbivorous wildlife likely represents one of the most important exposure pathways. The tundra vole is selected as the representative species for evaluating exposure for terrestrial wildlife. Tundra voles are highly herbivorous, and have small home ranges, which increases the realism of a scenario where receptors are exposed to a maximum food concentration in contrast to a wider ranging receptor such as the caribou, which may integrate exposure over larger spatial areas with varying chemical concentrations in food. Exposure parameters for the tundra vole used in the screening models are presented in Table 3-29. Although voles will consume a variety of plant types, for the purpose of this screening assessment, chemical concentration data for moss were used, as this food item has been analyzed for the broadest range of chemicals, and for those chemicals that have been measured in more than one plant type (i.e., lead, zinc, cadmium), the maximum concentrations in moss are higher than the maximum concentrations in other species (Exponent 2002a). Maximum chemical concentrations in tundra soils were also used as a measure of potential exposure via incidental soil ingestion, although the maximum soil and moss concentrations are not necessarily co-located for any chemical.

The results of exposure modeling for the tundra vole are shown in Table 3-30. All chemicals for which hazard quotients can be calculated have hazard quotients exceeding 1.0, except copper, fluoride, nickel, strontium, and tin. Fluoride data for moss were not available, and thus the hazard quotient for fluoride reflects exposure of voles to fluoride in tundra soil only. Appropriate TRVs have not been determined for iron and silver; therefore, hazard quotients cannot be determined for these chemicals. Water ingestion was not included in the exposure models for tundra voles, but because water ingestion is a minor route of exposure relative to food or soil ingestion, this exclusion is unlikely to alter the results of the screening, especially since most chemicals already have hazard quotients much greater than 1.0. For comparison purposes, similar hazard quotient calculations were performed using reference site data (Table 3-30). Only five chemicals had hazard quotients greater than 1.0 in the reference area: aluminum, barium, cobalt, manganese, and vanadium. In all cases except manganese, the reference area hazard quotient is substantially lower than the maximum site hazard quotient,

indicating that there are potentially incremental risks to receptors due to exposure to these four chemicals at the site. However, because the hazard quotient for manganese is approximately the same at the site (2.2) and at the reference area (2.1), there does not appear to be incremental risk associated with exposure to manganese at the site.

3.5.6.2 Piscivorous Wildlife

For aquatic habitats, chemical data are available for fish (Dolly Varden) in several streams that are crossed by the DMTS haul road, including Aufeis Creek, Omikviorok River, and Anxiety Ridge Creek. Chemical analyses of fish tissue samples are limited to four chemicals: cadmium, lead, selenium, and zinc, so only these four chemicals can be analyzed in the screening models. Fish data are used to model exposure to two piscivorous receptors: red-throated loon and river otter. Exposure parameters for these two receptors that are used in the screening models are presented in Table 3-29. For the purpose of this screening assessment, the maximum chemical concentration from any of the three creeks was used to calculate exposure for fish-eating wildlife.

The results of the exposure assessment for river otter and common loon are shown in Tables 3-31 and 3-32, respectively. For river otter, all hazard quotients are less than or equal to 1.0, while for loons, hazard quotients for lead, zinc, and cadmium are less than 1.0, but the selenium hazard quotient is 1.2 based on fish data from Aufeis Creek. Although the selenium hazard quotient equals 1.0 for river otter and slightly exceeds 1.0 for loons, recent fish sampling conducted by Ott and Morris (2004) indicates that the selenium concentrations in Dolly Varden from Aufeis Creek were similar to concentrations measured in fish from a creek in another mineralized area elsewhere in Alaska (Greens Creek). Thus, there does not appear to be any more incremental risk to river otters or loons from exposure to selenium at the site than at another mineralized stream in Alaska. Overall, results of the screening exposure models indicate a low likelihood of unacceptable risk to piscivorous wildlife from exposure to cadmium, lead, selenium, and zinc.

3.5.6.3 Invertivorous Wildlife

Effects to benthic invertivores that may forage in freshwater or coastal marine habitats cannot be assessed directly, as no data have been collected on chemical concentrations in benthic invertebrates. However, chemical data are available for sediment in streams that are crossed by the DMTS road, including New Heart Creek, Aufeis Creek, Omikviorok River, and Anxiety Ridge Creek, as well as in tundra ponds and coastal lagoons. For screening purposes, however, in exposure models it will be assumed that the maximum chemical concentrations in invertebrates inhabiting creeks, tundra ponds, or coastal lagoons are equal to the maximum sediment concentrations. This is a conservative assumption, because as noted in a review of biota-sediment accumulation factors (Bechtel Jacobs 1998), the median accumulation factor for arsenic, cadmium, chromium, nickel, and zinc are all less than 1, indicating that invertebrates do not accumulate these chemicals to levels greater than those measured in associated sediments. For copper, mercury, and zinc, the median BSAF was between 1 and 2, indicating that while these chemicals could be accumulated to levels greater than the associated sediment, the uptake would still be less than two-fold. A similar approach is used for estimating background risk

based on maximum chemical concentrations measured in reference creeks, ponds, and lagoons. Estimated benthic invertebrate concentrations are used to model exposure to the common snipe, which is selected as the representative freshwater invertivorous species (creeks and tundra ponds) and the black-bellied plover, which is selected as the representative marine invertivorous species (coastal lagoons). Exposure parameters for these receptors are shown in Table 3-29.

The results of the exposure assessment for avian invertivores are shown in Tables 3-33 through 3-35. Appropriate avian TRVs have not been determined for five chemicals (antimony, cobalt, iron, silver, and strontium); therefore, hazard quotients cannot be calculated for these chemicals. In creeks and streams traversed by the haul road, only four chemicals have hazard quotients exceeding 1.0 using the conservative estimate of benthic invertebrate tissue concentrations, specifically aluminum, barium, chromium, and lead (Table 3-33). However, hazard quotients for aluminum, barium, and chromium also exceed 1.0 in the reference creek, and the site and reference hazard quotients differ by less than two-fold. The selenium hazard quotient was equal to 1.0 in creeks and streams. Nine chemicals had hazard quotients exceeding 1.0 in site tundra ponds: aluminum, barium, cadmium, chromium, lead, mercury, selenium, thallium, and zinc (Table 3-34). However, of these chemicals, the hazard quotients for aluminum, barium, and chromium are less than those calculated at the reference lagoons, while selenium is less than two-fold greater than the corresponding reference area hazard quotient. Five chemicals had hazard quotients exceeding 1.0 in coastal lagoons: aluminum, barium, chromium, lead, and zinc (Table 3-35). However, all of these chemicals except zinc also had hazard quotients equal to or exceeding 1.0 in the reference lagoons, and for all except lead and zinc, the hazard quotient in site lagoons was less than two-fold greater than the reference area hazard quotient.

3.6 Selection of Ecological CoPCs

Chemical concentrations in environmental media were compared to various sets of ecological screening benchmarks as described in Section 3.5 and also to relevant reference area concentrations as described in Section 3.2.8. The purpose of this screening was to eliminate from further consideration those chemicals that are unlikely to have the potential for producing significant ecological effects while retaining those chemicals where such likelihood cannot be eliminated and where further evaluation is required. In this way, this approach helps to focus the ecological risk assessment (ERA) on those chemicals and exposure pathways where the potential for adverse ecological effects is greatest. In Sections 3.6.1 and 3.6.2, a tiered approach incorporating screening benchmark and reference data comparisons is applied to select CoPCs for plant, invertebrate, and fish communities and to eliminate from further consideration those chemicals that are unlikely to result in adverse ecological effects. In Section 3.6.3, results of the screening-level risk calculations are used to select CoPCs for wildlife.

3.6.1 Media Screening Evaluations

The environmental media screening evaluation uses a four-tiered approach to identify which chemicals should be retained as CoPCs and which ones can be eliminated from further consideration. In the first tier, maximum chemical concentrations in each environmental medium are compared with the lowest available screening benchmarks. In the second tier,

maximum concentrations are compared with other technically appropriate screening benchmarks. In the third and fourth tiers, frequency of benchmark exceedance is evaluated, and statistical comparison with reference area concentrations is performed. At each tier, chemicals that pass (i.e., maximum concentrations are lower than a screening value or not significantly different from reference concentrations) are dropped from the evaluation, while chemicals that fail the comparison are carried forward to the next tier. Chemicals that are undetected in all samples are eliminated if concentrations are less than screening values when their value is expressed as one-half of the detection limit, but are retained otherwise. Chemicals with no appropriate screening benchmarks in a specific medium are carried forward to the next tier.

3.6.1.1 First Tier Media Screening

In the first tier, media concentrations are screened against the following benchmarks:

- Chemical concentrations in tundra soils compared with the lowest ORNL benchmark based on effects to terrestrial plants, earthworms, or microbial heterotrophs
- Chemical concentrations in freshwater pond and stream sediment compared with TECs
- Chemical concentrations in marine and lagoon sediment compared with ERLs
- Chemical concentrations in freshwater and marine water compared with the CCC from the AWQC, with appropriate hardness adjustments applied when necessary for freshwater samples.

Results of this first tier of the screening comparison are presented in Table 3-36. No media or habitats were screened out completely on the basis of this screening comparison, although in general more chemicals were screened out in water than in sediment or tundra soil. Several undetected chemicals in water were screened out because their concentrations, expressed as one-half of the detection limits, were less than screening values (i.e., arsenic, chromium, mercury, and silver in stream water; mercury, selenium, and silver in tundra pond water; mercury in lagoon water; and chromium, mercury, and zinc in marine water).

3.6.1.2 Second Tier Media Screening

Multiple screening benchmarks are available for various media, especially for sediments. Therefore, in the second tier, chemical concentrations in sediments were also compared against SQS (marine/lagoon) or NEC values (pond/stream). The following briefly describes the technical basis of these screening benchmarks and their appropriateness for use in this screening assessment. The SQS were developed specifically to protect marine benthic macroinvertebrates and are technically appropriate for evaluations in northern nearshore environments. They were developed based on three kinds of standardized sediment toxicity tests and *in situ* evaluations of benthic communities. The SQS is established as the lowest no-effects level observed in any test, thus the most protective threshold is used to set the SQS for each chemical. In addition, SQS

were developed using a variety of quality assurance and data assessment steps designed to ensure that the results are valid and useful. In contrast, ERLs have several technical problems that limit their utility as screening benchmarks. First, they are based on many different investigations with many different biological endpoints, different sets of chemical analytes and widely varying laboratory and interpretive methods, and with no consistent quality assurance procedures applied. Additionally, because the ERL approach discards all no-effects data, the values cannot possibly be related (except coincidentally) to the no-effects level for any chemical. Instead, the values are simply a subjectively chosen percentile (the 10th percentile) of the distribution of concentrations in samples with some kind of biological effect. In addition, benthic communities in the Puget Sound area of Washington State, where the SQS were developed, are very similar to those found at the Red Dog port site, whereas the datasets used to develop the ERLs are not all biologically similar to northern Alaska. For these reasons, the SQS are considered technically stronger and more relevant to marine benthic communities in Alaska, while still being protective for screening purposes.

The method used to develop freshwater sediment NEC values is similar to the approach described above for derivation of SQS. The method, as described by Ingersoll et al. (1996) considers only the results of toxicity tests from field-collected sediment to identify concentrations of sediment contaminants above which adverse effects are always observed. It is assumed that if a field-collected sediment sample does not elicit a toxic response, it holds that individual contaminant concentrations in that sample are "safe." The NEC is thus the highest "safe" concentration observed across all field-collected samples evaluated. Alternatively, TEC values are derived using the same approach as described above for the ERL derivation, and suffer from the same technical limitations as described for those benchmarks.

In consideration of the technical issues associated with various sediment screening benchmarks, a second tier evaluation of sediment concentrations was done by comparing maximum concentrations against SQS or NEC values for all detected chemicals that failed the screening against ERL or TEC values, as well as undetected chemicals, and those without ERL or TEC screening benchmarks. The results of these comparisons are shown in Table 3-37. Use of these technically appropriate screening benchmarks results in a reduction in the number of chemicals that fail screening in marine, lagoon, pond, and stream sediment, although at least one chemical still exceeds its screening value in each environment.

The following chemicals were screened out at this second tier after having previously failed at the first tier:

- Arsenic, cadmium, and zinc in stream sediment
- Copper in tundra pond sediment
- Arsenic, lead, and nickel in lagoon sediment
- Arsenic, copper, nickel, and silver in marine sediment.

As there are no technically appropriate alternative screening values for saltwater, freshwater, or tundra soils, there is no further reduction of the CoPC list in these media relative to the results

presented in Table 3-36. However, it is important to note the limitations of the ORNL soil screening benchmarks for identifying CoPCs for the tundra plant and soil fauna communities at the site. The technical approach used to develop the ORNL benchmarks is similar to the methods used for deriving sediment ERL values (Long et al. 1995). The technical problems associated with ERL derivation that are discussed above are also relevant to the derivation of ORNL benchmarks. In addition, for some chemicals, information suitable for calculating soil screening values was limited to one or a few studies, and the authors expressed low confidence in these benchmarks (Efroymson et al. 1997a,b).

Toxicity tests used to develop the ORNL plant benchmarks were conducted predominantly in natural agricultural soils, such as loams and sands, which typically ranged from 1 to 20 percent organic matter (Efroymson et al. 1997a). The natural and artificial substrates used in earthworm and microbial toxicity tests also had low organic carbon contents (Efroymson et al. 1997b). Often, soluble (and highly bioavailable) forms of chemicals were added to soils in these tests, while most metals in natural soils exist in forms that are poorly bioavailable (Efroymson et al. 1997a,b). In particular, soils that are high in organic matter, such as the tundra soils found in the study area, strongly bind most metals, making them less available to biological organisms (McBride 1994). Therefore, the ORNL benchmarks are considered to be very conservative for a tundra environment. Furthermore, the phytotoxicity studies primarily examined effects on agricultural crop plants and tree species and may not be relevant for tundra vegetation. The tundra soil screening against ORNL earthworm benchmarks was included to represent terrestrial invertebrates at the site; earthworms would not be expected to occur in this arctic environment.

3.6.1.3 Third Tier Media Screening

For some chemicals, exceedance of screening benchmarks can be triggered by one or a few samples that exceed the values. In such cases, the spatial distribution of screening value exceedances may be limited relative to the spatial extent of the area under evaluation and is therefore not indicative of a widespread likelihood of risk across the assessment area. To account for these isolated exceedances of benchmarks, chemicals failing the screening tier summarized in Table 3-37 were examined in a third tier to determine the frequency of exceedance of the relevant benchmarks. For purposes of this screening assessment, it is considered that if a chemical exceeds its criterion in less than 10 percent of the samples analyzed for any given medium, then that chemical can be eliminated from further consideration as a CoPC for that specific medium. The results of this screening evaluation are presented in Table 3-38. The following chemicals were screened out at this third tier after having previously failed at the second tier:

- Lead in stream sediment
- Cadmium, iron, lead, and zinc in stream water
- Cadmium in lagoon sediment
- Zinc in lagoon water

- Cadmium, lead, mercury, and zinc in marine sediment
- Copper in marine water.

A few other chemicals only exceeded their screening benchmarks at a single location, but because of the small number of samples analyzed, the exceedance rate was greater than 10 percent, and these chemicals are retained in the screening process. These chemicals are mercury and nickel in pond sediments (1 of 5 exceedances each) and zinc in pond water (1 of 5).

3.6.1.4 Fourth Tier Media Screening

As a final comparison, all chemicals remaining after the third tier were statistically compared against chemical concentrations at the reference area. The rationale for this comparison is that even if chemicals exceed screening values, the likelihood of incremental risk to receptors from these chemicals is minimal if concentrations are not significantly different from levels receptors would be exposed to if they inhabited or were foraging in locations other than the study area. Additionally, for some chemicals without appropriate screening benchmarks, a comparison to reference concentrations can be used to eliminate them from further evaluation, as again, the incremental risk from exposure to these chemicals should be minimal, even though a benchmark comparison cannot be performed. The results of this comparison are shown in Table 3-39. The following chemicals were screened out at this fourth tier after having previously failed at the third tier:

- Aluminum, chromium, cobalt, iron, manganese, and nickel in tundra soil
- Barium, tin, and vanadium in stream sediment
- Aluminum, barium, cobalt, manganese, and strontium in stream water
- Barium, nickel, selenium, strontium, thallium, and vanadium in tundra pond sediment
- Barium, copper, fluoride, iron, lead, molybdenum, strontium, and vanadium in tundra pond water
- Aluminum, antimony, barium, cobalt, iron, molybdenum, selenium, strontium, thallium, and vanadium in lagoon sediment
- Aluminum, arsenic, barium, cobalt, iron, manganese, nickel, strontium, thallium, and vanadium in lagoon water
- Aluminum, cobalt, fluoride, iron, manganese, molybdenum, thallium, and vanadium in marine sediment
- Aluminum, antimony, barium, cobalt, fluoride, iron, manganese, molybdenum, and strontium in marine water.

3.6.2 Summary of Media Screening and CoPC Selection

The ecological screening process for chemicals in environmental media used a tiered approach that compared chemical concentrations against a series of ecological benchmarks and reference area concentrations to eliminate from further consideration those chemicals that do not pose a significant risk and to identify chemicals where further evaluation of ecological risks are required. Based on this evaluation, a final set of CoPCs for the ERA is identified, as shown in Table 3-40. The final set of CoPCs consists of two categories of chemicals: 1) chemicals that failed the screening based on comparisons against ecological benchmarks and were not screened out based on comparisons with reference concentrations, and 2) chemicals that lack appropriate screening benchmarks and were not screened out based on comparisons with reference concentrations. The potential for adverse effects due to the second group of chemicals is difficult to determine, because in the absence of appropriate screening benchmarks, the ecological relevance of concentrations that are elevated relative to the reference area cannot be determined. In some cases, these chemicals co-occur with other chemicals that have concentrations exceeding relevant benchmarks, which can make attribution of potential effects to chemicals without benchmarks problematic. While such chemicals are retained as CoPCs for the baseline assessment, risk characterization may be limited to narrative discussion of their potential to cause adverse effects as a component of the uncertainty assessment. The following sections briefly summarize the CoPCs identified in each habitat.

3.6.2.1 CoPCs in Terrestrial Tundra Habitats

Thirteen chemicals in tundra soil failed the screening based on comparisons against benchmarks and reference area concentrations and are retained as CoPCs for the baseline ERA. These chemicals include antimony, arsenic, barium, cadmium, copper, lead, mercury, molybdenum, selenium, silver, thallium, vanadium, and zinc. Strontium, which lacks an appropriate soil screening benchmark, is elevated in tundra soils at the site relative to the reference area and is retained on this basis.

3.6.2.2 CoPCs in Stream Habitats

Only nickel in stream sediment failed the screening based on comparisons with benchmarks and reference area concentrations and is retained as a CoPC for the baseline ERA. Eight other chemicals (antimony, cobalt, fluoride, molybdenum, selenium, silver, strontium, and thallium) lack appropriate sediment screening benchmarks and were not screened out based on comparisons with reference area concentrations, and are retained on this basis.

No chemical in stream water failed the screening based on comparisons with benchmarks and reference concentrations. However, six chemicals that lack benchmarks (antimony, fluoride, molybdenum, thallium, tin, and vanadium) were not screened out based on comparisons with reference stream data and are retained on this basis.

3.6.2.3 CoPCs in Tundra Pond Habitats

Four chemicals in tundra pond sediment (cadmium, lead, mercury, and zinc) failed the screening based on comparisons with benchmarks and reference area concentrations and are retained as

CoPCs for the baseline ERA, as are six other chemicals (antimony, cobalt, fluoride, molybdenum, silver, and tin) that lack relevant sediment screening benchmarks and were not screened out based on statistical comparisons with reference data.

Aluminum, cadmium, and zinc in pond water failed the screening based on comparisons with AWQC and were not screened out based on comparisons with reference area concentrations; these chemicals are retained as CoPCs for the baseline ERA. Five additional chemicals (antimony, cobalt, manganese, thallium, and tin) that lack AWQC and were not screened out based on comparisons with reference concentrations are retained on this basis.

3.6.2.4 CoPCs in Coastal Lagoon Habitats

Zinc is the only chemical in lagoon sediment that failed the screening based on comparisons with benchmarks and reference area concentrations, and it is retained as a CoPC for the baseline ERA. Three chemicals in lagoon sediments (fluoride, manganese, and tin) and four chemicals in lagoon water (antimony, fluoride, molybdenum, and tin) lack appropriate screening benchmarks and were not screened out based on comparisons with reference area concentrations.

3.6.2.5 CoPCs in Marine Habitats

No chemical in marine sediment or marine water failed the screening based on comparisons with benchmarks and reference area concentrations. The marine environment is the only habitat at the site where all detected chemicals for which there are appropriate screening benchmarks passed the media screening on the basis of benchmark comparisons. Five chemicals in marine sediment that lack appropriate screening benchmarks (antimony, barium, selenium, strontium, and tin) were not screened out based on comparisons with reference data, and the potential for adverse ecological effects as a result of exposure to these chemicals will be addressed narratively in the baseline ERA.

Three chemicals in marine water that lack appropriate benchmarks (thallium, tin, and vanadium), and nickel, which was undetected in all samples but exceeded the CCC at one-half the maximum detection limit, did not screen out based on comparisons with reference area concentrations, and the potential for adverse ecological effects due to these chemicals will also be addressed narratively in the baseline risk assessment.

3.6.3 Summary of Wildlife Screening and CoPC Selection

Food web models were constructed to evaluate exposure for representative terrestrial and aquatic receptors using site-specific data and conservative exposure assumptions. Exposure estimates were compared to no-effect level TRVs to calculate hazard quotients. The tundra vole was chosen as the representative terrestrial herbivore; the river otter and common loon as representative aquatic piscivores; and the common snipe as the representative aquatic invertivore.

Exposure models for the tundra vole indicated that 14 chemicals had hazard quotients exceeding 1.0, and thus cannot be screened out from further evaluation in the terrestrial environment. These chemicals are aluminum, antimony, arsenic, barium, cadmium, chromium, cobalt, lead, mercury, molybdenum, selenium, thallium, vanadium, and zinc. Manganese also had a hazard quotient exceeding 1.0, but because the hazard quotient calculated using reference area data was approximately equal to the site-specific value, there does not appear to be any incremental risk associated with this chemical at the site and it is not retained as a CoPC. Although fluoride data were not available for moss, fluoride was undetected in all but one of the tundra soil samples, and the hazard quotient calculated from exposure to the maximum fluoride concentration in tundra soil was very low (0.00061). Fluoride concentrations in moss would have to be about 50-fold higher than those in tundra soil for the total daily exposure to approach the TRV. Therefore, the likelihood of adverse effects from fluoride exposure is negligible, and fluoride is not retained as a CoPC for terrestrial herbivores. The 14 chemicals that are retained by the screening exercise will be evaluated in the baseline risk assessment by quantitatively evaluating risk to all terrestrial avian and mammalian herbivores in food web models. Because appropriate TRVs were not determined for iron and silver, these chemicals cannot be screened out, but they also cannot be evaluated quantitatively in exposure models. These two chemicals will therefore be evaluated qualitatively in the baseline risk assessment, where the likelihood of risk from these chemicals will be discussed relative to risk from chemicals for which derivation of numeric hazard quotients is possible. Data gaps currently exist for chemicals concentrations in prey of terrestrial carnivores and terrestrial insectivores, and screening cannot be performed for these receptors. Therefore, the same suite of chemicals identified as CoPCs for terrestrial herbivores will also be evaluated for risk to carnivores and insectivores, by collection of appropriate prey species and analysis for chemicals concentrations.

Exposure models for piscivorous wildlife using freshwater fish data indicated that the likelihood of risk from exposure to cadmium, lead, selenium, and zinc is low, and further evaluation of these metals is not required. Data limitations prevent screening of additional chemicals, as no other metals have to date been analyzed in Dolly Varden. However, the low hazard quotients for lead, zinc, and cadmium—given their relative abundance in the ore concentrates—suggest that risk from other metals is likely to be as low or lower than estimates for these three metals. Ott and Morris (2004) have proposed discontinuing annual sampling of Dolly Varden in Aufeis Creek and Omikviorok River in favor of sampling focused on streams near the mine, because metals concentrations in fish from these two creeks are low compared to sites near the mine, and concentrations are similar in fish upstream and downstream of the DMTS road. No analysis of metals concentrations has been done for marine fish inhabiting the coastal lagoons or nearshore marine habitats. However, because of the physiological mechanisms that fish use for osmoregulation in brackish and saltwater environments, uptake of metals is generally lower in marine fishes than in freshwater fishes (Hamelink et al. 1994). For these reasons, the results for freshwater piscivores are considered protective of marine piscivores, and the likelihood of effects to marine receptors is also considered to be low. Additionally, no CoPCs were identified in the marine environment based on screening and reference comparisons, suggesting that the likelihood of adverse effects to ecological receptors in that habitat is minimal. Overall, results of the screening assessment indicate that the likelihood of adverse effects to piscivorous wildlife from exposure to metals is low and no further evaluation of risk to these receptors is warranted.

Exposure models for benthivorous birds indicate that lead may potentially cause adverse effects in freshwater creeks and streams, as the hazard quotient was greater than 1.0 in Anxiety Ridge Creek, although not in the other three waterbodies that were evaluated. The hazard quotient for selenium equaled 1.0, but given the conservative nature of the food-web models, the likelihood that this indicates a significant adverse effect is considered minimal. Although hazard quotients for aluminum, barium, and chromium also exceed 1.0, the same chemicals also had hazard quotients exceeding 1.0 based on reference creek sediment concentrations. In some cases, the hazard quotients calculated for site creeks were less than the reference area estimates, and even in cases where they were higher, the difference was less than two-fold, indicating that the incremental risk due to exposure to these chemicals at the site was minimal. Therefore, these three chemicals are not retained as CoPCs for benthic invertivores in freshwater creeks and streams. In tundra ponds, five chemicals had hazard quotients exceeding 1.0, and these hazard quotients were also more than two-fold higher than the corresponding reference pond hazard quotients. These five chemicals, cadmium, lead, mercury, thallium, and zinc, are retained as CoPCs for benthic invertivores in tundra ponds. In coastal lagoons, lead and zinc had hazard quotients greater than 1.0 and these hazard quotients were more than two-fold higher than the corresponding reference hazard quotient. Therefore, these two chemicals are retained as CoPCs for benthic invertivores in coastal lagoons. Because appropriate TRVs were not determined for five chemicals (antimony, cobalt, iron, silver, and strontium), these chemicals cannot be screened out as CoPCs for invertivores in streams, tundra ponds, and coastal lagoons, and they also cannot be evaluated quantitatively in exposure models. These five chemicals will therefore be evaluated qualitatively in the baseline risk assessment, where the likelihood of risk from these chemicals will be discussed relative to risk from chemicals for which derivation of numeric hazard quotients is possible.

Currently no data exist to evaluate potential effects on herbivorous wildlife that may feed on aquatic plants in freshwater or coastal lagoon habitats, although, in general, uptake by aquatic plants should be low and cumulative aerial deposition may not be as great an exposure pathway as it is for terrestrial plants near the DMTS road. However, to address this data gap, plants will be collected from freshwater creeks, tundra ponds, and coastal lagoons and analyzed for the same suite of chemicals that were identified as CoPCs for terrestrial herbivores.

In summary, CoPCs that will be evaluated in quantitative food-web models in the baseline risk assessment are:

- For terrestrial herbivores, terrestrial insectivores, terrestrial carnivores, and aquatic herbivores: aluminum, antimony, arsenic, barium, cadmium, chromium, cobalt, lead, mercury, molybdenum, selenium, thallium, vanadium, and zinc
- For avian invertivores foraging in freshwater streams and creeks: lead
- For avian invertivores foraging in tundra ponds: cadmium, lead, mercury, thallium, and zinc
- For avian invertivores foraging in coastal lagoons: lead and zinc.

4 Human Health Risk Assessment

The purpose of the HHRA is to evaluate the likelihood that health effects could occur in people who come into contact with the CoPCs associated with the DMTS road corridor. The DMTS HHRA uses standard procedures developed by EPA and DEC adapted, when appropriate, to the specific conditions of the site. The first two steps of the HHRA, development of a preliminary CSM and the CoPC screening, were described in Sections 2.3 and 3.3, respectively. The following sections describe the methodology that will be used to conduct the remainder of the DMTS HHRA. Section 4.1 describes refinements to the preliminary CSM based on the results of the CoPC screening. Section 4.2 presents the methodology that will be used in the exposure assessment, which will quantify the amount of exposure to site CoPCs that could potentially occur. Section 4.3, the toxicity assessment, summarizes current scientific knowledge regarding the toxicity of site CoPCs. Section 4.4, the risk characterization, describes how the results of the exposure and toxicity assessments will be used to derive risk estimates for the site.

4.1 Refined Conceptual Site Model

Based on the results of the human health CoPC screening in Section 3.3, there are four CoPCs in the terrestrial environment (barium, cadmium, lead, and zinc), no CoPCs in the freshwater environment, and no CoPCs in the marine environment. The refined CSM (Figure 4-1) reflects the screening process. Thus, potential exposures related to the freshwater and marine environments will not be evaluated quantitatively in the risk assessment because the conservative screening process indicates that there is little or no risk related to site activities in these environments. The exposure pathways in the terrestrial environment remain unchanged from the preliminary CSM. Thus, risks will be quantitatively evaluated for soil and dust ingestion, and for subsistence food consumption in the terrestrial environment.

4.2 Exposure Assessment

In an HHRA, exposure assessment is the process of identifying human populations that could potentially contact site-related CoPCs and estimating the magnitude, frequency, duration, and route(s) of potential exposures. An exposure pathway describes a chemical's transport from its source to a potentially exposed individual and must include a source, transport mechanism, receptor, and point of entry into the body. Only when each of these elements is present can an exposure pathway be complete, and only complete exposure pathways have the potential to result in a health risk. Potential exposures associated with the CoPCs identified at the site are evaluated by identifying current and potential future uses of the property, those populations that could be exposed to the chemicals (i.e., the receptors), and the manner in which they may be exposed (i.e., the exposure pathway). The relevant exposure pathways are described in the CSM section above.

This section describes the methodology used to quantify exposure for the complete exposure pathways identified in the CSM. Consistent with guidance from both DEC and EPA, reasonable

maximum exposure (RME) estimates will be applied for all complete exposure pathways. Exposure and risk estimates will be derived using deterministic methodology. Because exposure assessment for lead differs from that of other metals, these methods are described separately.

4.2.1 Exposure Concentrations

EPA guidance (U.S. EPA 1989, 1992b, 2002b) indicates that exposure concentrations used in risk assessment calculations should be either the 95 percent upper confidence limit (UCL) on the mean concentration or the maximum site concentration, whichever is lower. EPA recommends the 95 percent UCL as an estimate of mean exposure concentration because of the uncertainty associated with estimating the true average exposure concentration at a site. For normally distributed data, EPA recommends calculating the UCL based on the Student's *t*-statistic. This can be calculated using the following equation:

$$UCL = \overline{X} + (t_{\alpha,n-1} \times s) \Big/ \sqrt{n}$$

where:

$\overline{X}$ = arithmetic mean of site concentration data

t = Student's t-value for a given α and n

α = one-sided confidence level (0.05 for the 95 percent UCL)

n = number of observations

s = standard deviation of site concentration data.

For lognormally distributed data, EPA recommends using either the Land method or the Chebyshev inequality method, depending on sample variance and the number of samples (U.S. EPA 2002b). A 95 percent UCL using the Land method is calculated using the following formula:

$$UCL = \exp\left(\overline{y} + \frac{S_y^2}{2} + \frac{S_y \times H}{\sqrt{n-1}}\right)$$

where:

exp = exponential function

$\overline{y}$ = average of the log-transformed data (y = ln(x))

S_y = standard deviation of the log-transformed data.

H = H statistic for a given confidence level, n, and S_y

n = number of observations.

A 95 percent UCL using the Chebyshev inequality method is calculated using the following formula:

$$UCL = \bar{\mu} + \sqrt{\left(\frac{1}{\alpha} - 1\right)\sigma_{\mu}^2}$$

where:

$\bar{\mu}$ = minimum-variance unbiased estimate (MVUE) of the population mean

α = one-sided confidence level (0.05 for the 95 percent UCL)

σ_{μ}^2 = MVUE of the variance of the population mean.

The MVUE of the population mean is calculated as follows:

$$\bar{\mu} = \exp(y) \times g_n \times (s_y^2/2)$$

where g denotes a function for which tables are available, as cited in U.S. EPA (2002b). All other parameters are as described above.

The MVUE of the variance of the population mean is calculated as follows:

$$o_{\mu}^2 = \exp(2y) \times \left(\left((g_n \times (s_y^2/2)) \right) - \left(g_n \times \left(\frac{n-2}{n-1} s_y^2 \right) \right) \right)$$

Site data will be evaluated using appropriate statistical distribution testing methods (e.g., the Kolmogorov-Smirnov test for normality or lognormality), and exposure concentrations calculated using the recommended statistical methods, as described above. U.S. EPA (2002b) describes several methods for handling data with distributions that cannot be identified as either normal or lognormal. In the event that a distribution cannot be identified for a particular data set, or is identified as something other than normal or lognormal, the appropriate methodology will be applied using EPA guidance.

Exposure concentrations for lead will be calculated using arithmetic means. As described below and in model guidance, the IEUBK and adult lead models are designed to be applied using average values as input. A geometric standard deviation (GSD) is then applied to account for variability.

4.2.2 Subsistence Use

This subsistence use receptor scenario addresses exposures that could potentially occur due to subsistence food consumption, and the incidental soil and dust ingestion that could occur while a person is engaged in subsistence hunting and harvesting activities. Exposure quantification methods are first described for lead and then for the remaining CoPCs.

4.2.2.1 Lead Exposure

Unlike the other CoPCs, lead exposure is evaluated by estimating its effect on increasing blood lead levels rather than by calculating a daily dose per body weight. EPA has developed two models for assessing lead exposure: the IEUBK model (U.S. EPA 1994) for assessing lead exposure in young children, and a simplified linear model for assessing exposure in older children and adults (EPA adult lead model; U.S. EPA 1996c). Both models predict steady-state chronic blood lead levels and incorporate health-protective assumptions about behavior. Because young children are much more sensitive to lead than adults, the adult lead model is based on potential impacts on the developing child (i.e., on the fetus) and the IEUBK model evaluates potential effects to the child following childhood intake of lead. The IEUBK model will be used to assess exposure to lead during subsistence hunting and gathering activities and in the subsistence diet. The IEUBK model provides a far more conservative assessment of risks than the adult lead model; therefore it is unnecessary to also apply the adult lead model for the subsistence use scenario. However, the adult lead model will be applied to assess workers' cumulative exposure to lead during occupational activities, in consuming subsistence foods, and during subsistence hunting and gathering activities.

The EPA IEUBK child lead model differs from the adult model in that the child model has inputs for lead exposure from a number of sources, including soil, diet, air, the maternal contribution *in utero*, and water.[7] The IEUBK model (Windows Version 1.0) will be used to assess lead exposure to the sensitive population (i.e., young children) under the subsistence use scenario. This model estimates a geometric mean blood lead level based on site exposure as well as other background sources. Like the adult model, a GSD is then applied to estimate upper percentile blood lead levels. The assumptions used in this model will be EPA defaults (U.S. EPA 1994), with the exception of soil concentrations, gastrointestinal absorption for soil, drinking water concentration, and dietary intake (Table 4-1).

Soil Lead—The soil lead concentration input to the model will be calculated using the arithmetic mean of lead concentrations collected from the port industrial areas, the road, and the road shoulder. As discussed previously, there is little bare soil in the tundra outside of the road and port, and people would come into relatively little contact with the inorganic soil underneath the tundra mat of decayed organic material. Although soil and dust exposure could also potentially occur by contacting dust on plant and animal surfaces, chemical concentrations in soil and dust away from the road and port would be considerably lower than on the road and port industrial area if those chemicals concentrations were related to fugitive dust. Thus, use of data only from the road and port industrial area will provide a conservative estimate of chemical exposure from soil and dust.

Gastrointestinal Absorption of Soil Lead—The default soil lead bioavailability input to the IEUBK model is 30 percent. However, U.S. EPA (1999b) guidance acknowledges that different forms of lead in soil will vary in their bioavailability. The lead present in Red Dog Mine concentrate is primarily galena (lead sulfide) (Arnold and Middaugh 2001; DEC et al. 2002). U.S. EPA (1999b) identifies galena as a form that is likely to have a bioavailability lower than

[7] The adult model adds in a background value for blood lead that would include all other exposures to lead from sources such as air, water, and diet, while the IEUBK model requires entry of all environmental lead data and does not include an input parameter for background blood lead.

the default value of 30 percent. The Alaska Division of Public Health published data from a National Toxicology Program (NTP) study examining bioavailability of lead in Red Dog concentrate (Arnold and Middaugh 2001; Arnold et al. 2003). The study was conducted using a standard NTP protocol, whereby juvenile (6 to 8 week-old) male Fisher 344 rats were fed diets supplemented with either Red Dog ore or soluble lead (i.e., lead acetate) in the same concentrations for 30 days. Red Dog ore was sieved to particle sizes less than 38 microns prior to diet supplementation. Blood lead levels, as well as other tissue lead concentrations, were determined at the end of the 30-day period.

As summarized in Table 4-2, Arnold and Middaugh (2001) reported relative bioavailability of lead in Red Dog ore ranging from 13.6 percent to 27 percent. Relative bioavailability is calculated by dividing the blood lead concentration after feeding the animal with lead from ore, by the blood lead concentration after feeding the animal with the same amount of soluble lead acetate. The IEUBK model requires absolute bioavailability as an input (U.S. EPA 1996c). Absolute bioavailability is the amount that enters the blood stream relative to the amount that is ingested. This is estimated by multiplying the absolute bioavailability of soluble lead acetate by the relative bioavailability of Red Dog ore lead. The IEUBK model assumes an absolute bioavailability of 50 percent for soluble lead. Thus, absolute bioavailability of Red Dog ore is calculated by multiplying the relative bioavailability of Red Dog ore by 0.5 (i.e., 50 percent). Absolute bioavailability of Red Dog ore ranged from 6.8 percent at 100 mg/kg to 13.5 percent at 10 mg/kg. The average absolute bioavailability in the study was 9.7 percent. For the DMTS risk assessment, risks will be evaluated using both the IEUBK model default bioavailability of 30 percent and the site-specific value of 9.7 percent.

The trend in the NTP study is for lower bioavailability with increasing lead concentrations (Arnold and Middaugh 2001). Thus, use of the average bioavailability across the range of concentrations investigated in the study, including those from the relatively low lead concentrations, is likely to provide a conservative estimate of bioavailability. It is notable that, despite elevations in soil lead along the DMTS, blood lead concentrations in residents of Kivalina and Noatak were found to be "very low" in the early 1990s by Arnold and Middaugh (2001). This suggests low bioavailability, low exposure, or both.

Drinking Water Lead—The default drinking water lead concentration input to the IEUBK model is 4 μg/L. However, site data indicate that water lead concentrations are significantly lower in the area. The site arithmetic mean stream surface water lead concentration of 0.7 μg/L will be used in the DMTS risk assessment.

Subsistence Food Lead—Model input for subsistence food lead intake will be estimated using a combination of subsistence food intake data for Kivalina and Noatak available from the DFG Community Profile Database (CPDB; DFG 2001), and tissue lead data for relevant food items. Derivation of subsistence food consumption rates for the risk assessment is described in detail in Section 4.2.2.3. The food items identified in the CSM as representative of subsistence use in the area that are relevant to the terrestrial environment are caribou, ptarmigan, berries, and sourdock. Average lead concentrations in these food items will be used as input to the IEUBK model.

4.2.2.2 Exposure Assumptions for Non-Lead Chemicals

Non-lead chemicals exposure will be evaluated by combining estimates of media (soil and food) intake with estimates of the chemicals concentrations in those media. As with lead, exposure to other chemicals in subsistence food and in soil and dust during subsistence hunting and gathering will be evaluated in children. However, because adults could potentially have a greater exposure (per kilogram body weight) in subsistence foods than children, adults will also be evaluated for exposure to non-lead chemicals.

Soil—Soil exposure to non-lead chemicals will be evaluated under the subsistence hunter and gatherer scenario using standard EPA equations and RME assumptions (U.S. EPA 1989, 1991). Exposure concentrations for soil will be calculated based on the lesser of the 95 percent UCL of the mean or the 95 percent UCL of the maximum detected concentration for each area.

The estimated daily intake for each chemical from soil will be estimated using the following equation:

$$\text{Intake } (\text{mg/kg-day}) = \frac{C_s \times 10^{-6} \times IR_s \times ED \times EF \times FI}{BW \times AT}$$

where:

C_s	=	chemical concentration in soil (mg/kg-dry weight)	= chemical-specific
10^{-6}	=	conversion of mg soil to kg soil	
IR_s	=	soil ingestion rate (mg/day)	= 200 for children = 50 for adults
ED	=	exposure duration (years)	= 6 for children = 30 for adults
EF	=	exposure frequency (days/year)	= 200 for arctic zone
FI	=	fractional intake from site (unitless)	= to be derived
BW	=	body weight (kg)	= 15 for children = 70 for adults
AT	=	averaging time (days)	= 5,475 for children = 10,950 for adults.

Subsistence Food—Exposure to non-lead chemicals in subsistence foods will be evaluated by combining estimates of daily intake of specific food items with estimated chemical concentrations in those items. The daily intake of chemicals from subsistence foods will be estimated using the following equation:

$$\text{Intake } (\text{mg/kg-day}) = \frac{C_f \times 10^{-3} \times CR_f \times ED \times EF \times FI}{BW \times AT}$$

where:

C_f	=	chemical concentration in food item (mg/kg-wet weight)	= chemical-specific
10^{-3}	=	conversion of g food to kg food	
CR_f	=	food item consumption rate (g/day)	= food-specific
ED	=	exposure duration (years)	= 6 for children = 30 for adults
EF	=	exposure frequency (days/year)	= 365
FI	=	fractional intake from site (unitless)	= to be derived
BW	=	body weight (kg)	= 15 for children = 70 for adults
AT	=	averaging time (days)	= 5,475 for children = 10,950 for adults.

The doses from all food items will then be summed to obtain an estimated subsistence food intake rate. The amount of subsistence consumption related to gathering along the DMTS will be considered through application of a fractional intake term. Fractional intake will be derived by estimating the area of the site relative to all subsistence hunting and gathering areas. The fractional intake is calculated as follows:

$$\text{Fractional Intake} = \frac{\text{area of site}}{\text{total area where subsistence hunting and gathering occurs}}$$

4.2.2.3 Review of Existing Subsistence Food Data

Neither EPA nor DEC provides specific instructions or input data for subsistence food consumption calculations in risk assessments. U.S. EPA (1997b) provides food consumption data for the U.S. general population and some specific subpopulations, but not for Native Alaskan subsistence groups. Whenever possible, food consumption data specific to the populations being evaluated should be used in a risk assessment.

The CPDB, developed by DFG (2001), provides information on subsistence fish and wildlife harvests in Alaska. The CPDB also contains a wide array of current socioeconomic data. Information is derived from more than a decade of research by the DFG Division of Subsistence, and from other sources. The database was developed from information collected during household interviews conducted between 1980 and 2000. During interviews, respondents were asked questions regarding the types and quantities of subsistence resources harvested and consumed during the past 12-month period. Individual household data were compiled and summarized by community. The numbers in the database represent a single year's harvest and use of subsistence resources from a complete seasonal cycle of fishing, hunting, gathering, and trapping activities. For some communities, data were collected during more than 1 year. Typically, however, only one of the years provides a complete record of subsistence use. The year with the complete record is identified in the CPDB as the most

representative year. Only data from the most representative year will be used in the DMTS risk assessment.

For the DMTS risk assessment, the CPDB was queried for subsistence food harvest and use in Kivalina and Noatak. The most representative data for Kivalina were collected in 1992 from 62 of the 72 households in the village (N=296 people). The most representative data for Noatak were collected in 1994 from 68 of the 84 households in the village (N=307 people). Estimated use for Kivalina and Noatak of the seven major categories of subsistence foods (i.e., land mammals, migratory birds, game birds, fish, marine invertebrates, marine mammals, and vegetation), along with use of major food items within those categories, are summarized in Table 4-3. Data from the two communities were averaged to derive subsistence food use rates for a typical user in the area.

As is typically the case in retrospective diet history surveys such as the CPDB (e.g., Rasanen 1979), when estimates of food consumption for all food items are summed, the resulting total food intake greatly overestimates actual food consumption. For example, as shown in Table 4-3, the total estimated food intake derived by summing the intake from each main category of subsistence food is 830 g/day, or more than 4,200 kcal per day. This intake greatly exceeds a person's energy needs (FDA, no date). For example, by FDA (2003) calculations, the caloric intake requirements of an active 70 kg adult are approximately 2,850 kcal per day, and 1,650 kcal/day for a 70 kg adult with low activity levels.

Nobmann and colleagues (1992) conducted a study on dietary intake in Native Alaskans from 10 communities throughout Alaska (including Kotzebue). Their methodology included the use of multiple 24-hour recall surveys, completed during five seasons over an 18-month period. This type of dietary assessment (i.e., the 24-recall) has been shown to accurately reflect dietary patterns (e.g., Witschi 1990). Nobmann et al. (1992) reported the typical caloric intake for native Alaskans as approximately 2,750 kcal per day for men and 1,950 kcal per day for women (Table 4-4; Nobmann et al. 1992). Caloric intake in the general U.S. population during that time period was approximately 2,550 kcal per day for men and 1,550 kcal per day for women (NHANES II, as reported in Nobmann et al. 1992). The Nobmann et al. (1992) data clearly illustrates the degree to which the CPDB database overestimates intake. In addition, the data provided by CPDB include only subsistence foods, whereas Nobmann et al. (1992) include all food consumption, including store-bought foods. Thus, the estimates of total average subsistence food consumption in Kivalina and Noatak given by CPDB likely overestimate actual food consumption by at least 2-fold.

In order to use the data provided by CPDB on the relative amounts of different food items consumed, the data must first be modified to account for actual caloric intake. Caloric intake-weighted subsistence food consumption rates were derived using the following methodology (and as summarized in Tables 4-3 and 4-4):

1. Total CPDB-derived subsistence food use for the area was calculated by summing the estimated CPDB-derived subsistence use rate for each of the seven major categories of subsistence foods. The estimate is 830 g/day (Table 4-3).

2. Food consumption estimates in the CPDB database are given in grams. The intake in grams must be converted to calories to derive caloric intake-weighted consumption rates. There are three components of food that provide calories (excluding alcohol): protein, fat, and carbohydrates. The caloric density (i.e., the kcal per gram) of a food depends on the relative amounts of these components in the food. Using data provided in Nobmann et al. (1992) on protein, fat, and carbohydrate intake in the Native Alaskan diet, the average caloric density was estimated by the following method (Table 4-4):

   - Multiply the grams intake from Nobmann et al. (1992) of each of the three components of the diet by their specific energy content (protein, 4 kcal/g; fat, 9 kcal/g; carbohydrate, 4 kcal/g [Merrill and Watt 1973]). There was no alcohol consumption reported (the only other dietary component that could provide energy).

   - Sum the caloric intake from protein, fat, and carbohydrate (2,689 kcal).

   - Divide the total caloric intake (2,689 kcal) by the total food intake in grams (526 g) to derive the average energy per gram of food (5.1 kcal/g).

3. The total CPDB-derived subsistence food use rate of 830 g/day was converted to caloric intake by multiplying it by 5.1 kcal/g. The estimate is 4,234 kcal/day (Table 4-3).

4. A caloric intake-weighting factor for adults of 0.65 was derived by dividing the total caloric intake from Nobmann et al. (1992) of approximately 2,750 kcal by the CPDB-derived caloric intake of 4,234 kcal (Table 4-3).

5. Caloric intake-weighted consumption rates for adults were derived by multiplying the CPDB-derived consumption rates by the caloric intake weighting factor (Table 4-3).

6. Caloric intake-weighted consumption rates for children (0 to 6 years old) were derived using the same methodology, but assuming a total caloric intake that is half that of adults (i.e., 1,375 kcal/day) (FDA 2003). There are no intake data available specific to Native Alaskan children.

Consumption rates are presented for all seven major subsistence food categories for the purposes of calculating the caloric intake-weighted consumption rates. However, only the consumption rates for the categories that have been identified in the work plan for evaluation in the risk assessment will be used. These consumption rates are based on the best available data relevant to the population of interest. They are considered to be conservative because they were derived under the assumption that all food intake comes from subsistence foods. Inclusion of non-subsistence food sources in their derivation would result in lower consumption rate estimates for subsistence foods.

4.2.3 Combined Occupational and Subsistence Use

The occupational and subsistence use receptor scenario addresses exposures to future hypothetical DMTS workers that could potentially occur through incidental soil ingestion and due to subsistence food consumption, and the incidental soil and dust ingestion that could occur while a person is engaged in subsistence hunting and harvesting activities. Exposure quantification methods are first described for lead and then for the remaining CoPCs.

4.2.3.1 Lead

Adults are the appropriate receptors for soil lead exposure for this receptor scenario because it is focused on combined workplace and subsistence exposures. Thus, the adult lead model, which is recommended for adults and older children, is the appropriate choice of a lead uptake model. The EPA adult lead model was developed based on Bowers et al. (1994) with some modifications to input assumptions, which generally make the model more conservative (i.e., the modifications result in higher predicted blood lead levels). Although this model has not been fully peer-reviewed or rigorously validated, EPA recommends it for assessing exposure of older age groups (adolescents to adults) to lead in soil. The model is used to evaluate lead exposure to the most sensitive subpopulation: the fetuses of pregnant women who work on the DMTS and engage in subsistence use in the area. As described above, although the DMTS has signage in place to limit exposure, the risk assessment assumes that exposure will occur.

The adult lead model is essentially an equation that estimates an average blood lead level based on additional exposure (above a baseline level) to lead in soil and air. A separate input in the equation for inhalation of lead from dust in the air is not necessary because the majority of airborne dust is not inhaled into areas of the lung where absorption of chemicals could occur. As described in Section 2.3.3 of the CSM, most inhaled dust only reaches the upper respiratory tract, where it is carried into the esophagus and ultimately ingested. Exposure from inhaled dust is included in the intake given by the soil ingestion rate. The equation is thus:

$$PbB_{central,\,adult} = PbB_0 + \frac{BKSF \times C_s \times IR_s \times EF_s \times AF_s \times FI_s}{AT}$$

where:

$PbB_{central,\,adult}$	=	geometric mean blood lead level for adults, central estimate (μg/dL)		
PbB_0	=	maternal baseline blood lead level (μg/dL)	=	1.53
BKSF	=	biokinetic slope factor (μg/dL per μg/day)	=	0.4
C_s	=	lead concentration in soil (μg /g-dry weight)	=	to be determined
IR_s	=	soil ingestion rate (g/day)	=	0.05
EF_s	=	exposure frequency (days/year)	=	200 for arctic zone
AF_s	=	absorption fraction (unitless)	=	0.12 and 0.039
FI_s	=	fractional intake of soil from site (unitless)	=	to be derived
AT	=	averaging time (days/year)	=	365.

The general formula can be modified to take into account lead intake from other sources, such as diet, as follows:

$$PbB_{central,\,adult} = PbB_0 + \frac{BKSF \times \left[\left(C_s \times IR_s \times EF_s \times AF_s \times FI_s\right) + \left(C_f \times CR_f \times EF_f \times AF_f \times FI_f\right)\right]}{AT}$$

where:

IR_s	=	food item consumption rate (g/day)	=	food specific
C_f	=	lead concentration in food item (μg /g-wet weight)	=	to be determined
EF_f	=	exposure frequency for subsistence food (days/year)	=	182.5
AF_f	=	absorption fraction from food (unitless)	=	0.2
FI_f	=	fractional intake of subsistence food from site (unitless)	=	to be derived.

Each food item will have its own concentration and consumption rate term. To predict a central tendency (geometric mean) blood lead level, all inputs should be central tendency (i.e., average) estimates, not reasonable maximums. To calculate the 95th percentile blood lead level, a GSD is applied:

$$PbB_{95,\,adult} = PbB_{central} \times GSD^{1.645}$$

where:

$PbB_{95,\,adult}$	=	95th percentile estimate of blood lead level for adults (μg/dL)
GSD	=	individual geometric standard deviation (unitless) = 2.11
1.645	=	95th percentile value for Student's t distribution.

Fetal blood lead levels are predicted on the basis of the EPA assumption that fetal blood lead levels at birth are 90 percent of the maternal blood lead level. Thus, the 95th percentile estimate fetal blood lead level is estimated as follows:

$$PbB_{95,\,fetal} = PbB_{95,\,adult} \times R$$

where:

$PbB_{95,\,fetal}$	=	95th percentile estimate of blood lead level for fetus (μg/dL)
R	=	fetal-to-maternal constant of proportionality = 0.9 (unitless).

An alternative method of separating soil lead and dust lead ingestion is necessary if there is more than one separate source with a different concentration and/or characteristic (e.g., if the contribution from lead-based paint is being assessed). For the DMTS, the source of soil lead and dust lead is the same. Thus, intakes of the two are combined, as recommended in the adult

lead model guidance (U.S. EPA 1996c). Furthermore, U.S. EPA (1996c) states that a soil ingestion rate of 50 g/day addresses both direct intake from soil and indirect intake through ingestion of dust, and that "no specific assumptions are needed about the fraction of soil intake that occurs through dust."

Site-specific modifications to the EPA default assumptions (U.S. EPA 1996c) are described below.

Baseline Blood Lead Level—U.S. EPA (2002a) reports updated values for baseline blood lead in U.S. females from 17 to 45 years of age. Although data are reported for different regions and ethnic groups, there are no data available for either the specific region or ethnic group relevant to this risk assessment. Therefore, the reported mean baseline blood lead of 1.53 μg/dL for all ethnic groups combined will be used as input for the adult lead model in the risk assessment.

Exposure Frequency to Soil—EPA recommends an exposure frequency for workers of 219 days per year for the adult lead model (U.S. EPA 1996c). However, this parameter must be modified to take into account the number of days without snow cover, as expressed in the DEC (2002) recommended residential exposure frequency for the arctic zone of 200 days/year. Because this scenario assumes that exposure is occurring while at work and while away from work, the residential exposure frequency of 200 days/year will be used as input for the adult lead model.

Gastrointestinal Absorption Fraction from Soil—As discussed in the section describing the IEUBK model, site-specific bioavailability data indicate that the form of lead at the site is less bioavailable than the default input to the models. Furthermore, U.S. EPA (1996c) acknowledges that lead is less bioavailable in adults than in children. The default absolute bioavailability of soluble lead in the IEUBK model is 50 percent, whereas the default bioavailability for soluble lead in the adult model is 20 percent, due to the reduced absorption of lead observed in adults relative to children. The relative bioavailability data for Red Dog ore reported by Arnold and Middaugh (2001) and summarized in Table 4-2 indicated lower bioavailability than the default values. These site-specific data can be applied to the adult lead model by multiplying the relative bioavailability of Red Dog ore by the default adult absolute bioavailability of soluble lead of 0.2 (i.e., 20 percent). The resulting absolute bioavailability of lead in Red Dog ore for adults ranges from 2.7 percent to 5.4 percent, with an average of 3.9 percent (i.e., 0.039). In the DMTS risk assessment, risks will be evaluated using both the adult lead model default bioavailability of 12 percent and the site-specific value of 3.9 percent.

Fractional Intake of Soil from Site—The fractional intake of soil (FI_s) from the site used in the risk assessment for the adult lead model will be time-weighted to account for the difference in fractional intake that will be assumed to occur during the two-thirds of the time that a person is on work rotation vs. the one-third of the time they are off work. Specifically, it will be assumed that while working, 100 percent of soil ingestion will occur at the site. When not at work, the fraction of soil ingested at the site will be equivalent to the fractional intake described in the subsistence user scenario. FI_s is described mathematically as follows:

$$FI_s = \left(1.0 \times \frac{2}{3}\right) + \left(FI \times \frac{1}{3}\right)$$
$$= 0.67 + \frac{FI}{3}$$

Exposure Frequency to Subsistence Food Consumption—The combined worker and subsistence use scenario must reflect the differences of subsistence food consumption during the two-thirds of the time that a person is at work and the one-third of the time they are not at work. For the DMTS risk assessment, it will be assumed that subsistence food consumption occurs during 100 percent of the 121.7 days/year (i.e., 1/3 × 365 days/year) that a person is off work. It is not possible to quantify the amount of subsistence food consumption that occurs while on a work shift, but it would be expected to be low because all meals are provided in the Red Dog cafeteria. For the DMTS risk assessment, it will be conservatively assumed that 25 percent of food intake during the 243.3 days/year (i.e., 2/3 × 365 days/year) while on a work shift is subsistence food, or the equivalent of 60.8 days/year on average. Thus, EF_f that will be used in the DMTS risk assessment is the combined exposure frequency while off work and while on a work shift of 182.5 days per year (i.e., 121.7 + 60.8).

Absorption Fraction of Lead from Subsistence Foods—In the IEUBK model guidance, U.S. EPA (1994) recommends use of the same lead absorption fraction for food and soluble lead (as in water). EPA provides no specific guidance on input assumptions for food intake in the adult lead model guidance. Therefore, an assumption analogous to that recommended in the IEUBK model guidance will be used in the DMTS risk assessment. The default absolute absorption assumed for soluble lead in the adult lead model is 0.2 (i.e., 20 percent) (U.S. EPA 1996c). Thus, an absorption fraction for lead in foods of 0.2 will be used in the adult lead model for the DMTS risk assessment.

Fractional Intake of Subsistence Foods from Site—The FI_f will be the same as the fractional intake used in the subsistence user scenario. Thus, FI_f will be derived by estimating the area potentially affected by fugitive dust deposition, relative to all subsistence hunting and gathering areas. Differences in subsistence food consumption while off work versus while on a work shift are addressed in the *Exposure Frequency to Subsistence Food Consumption* (EF_f) section described above.

Geometric Standard Deviation—The GSD is an estimation of variation in blood lead around the geometric mean, and is used to estimate upper percentile blood lead levels for an individual and predict the probability of an individual exceeding a given blood lead level (target risk goal).

U.S. EPA (2002a) reports updated values for blood lead GSD in U.S. females from 17 to 45 years of age. Although data are reported for different regions and ethnic groups, there are no data available for either the specific region or ethnic group relevant to this risk assessment. Therefore, the combined GSD of 2.11 for all ethnic groups in all regions will be used as input for the adult lead model in the risk assessment.

4.2.3.2 Non-Lead Chemicals

Non-lead chemicals exposure will be evaluated by combining estimates of media (soil and food) intake with estimates of the chemical concentrations in those media (i.e., the exposure point concentrations calculated as described in Section 4.2.1).

Soil—Soil exposure to non-lead chemicals for the combined worker and subsistence hunter and gatherer scenario will be evaluated using standard EPA equations and RME assumptions (U.S. EPA 1989, 1991). Exposure concentrations for soil will be calculated based on the lesser of the 95 percent UCL of the mean or the 95 percent UCL of the maximum detected concentration for each area.

The daily intake for each chemical from soil will be estimated using the following equation:

$$\text{Intake } (\text{mg/kg-day}) = \frac{C_s \times 10^{-6} \times IR_s \times ED \times EF \times FI_s}{BW \times AT}$$

where:

C_s	=	chemical concentration in soil (mg/kg-dry weight)	=	chemical-specific
10^{-6}	=	conversion of mg soil to kg soil		
IR_s	=	soil ingestion rate (mg/day)	=	50
ED	=	exposure duration (years)	=	30
EF	=	exposure frequency (days/year)	=	200 for arctic zone
FI_s	=	fractional intake of soil from site (unitless)	=	to be derived
BW	=	body weight (kg)	=	70
AT	=	averaging time (days)	=	10,950.

All input assumptions for soil exposure in this scenario will be the same as described for the adult subsistence user scenario (in Section 4.2.2.2), with the exception of FI_s. FI_s will account for the fact that two-thirds of the time (i.e., while at work) essentially all soil exposure will be assumed to take place at the site (i.e., during work that is assumed to occur solely on and near the road), and one-third of the time (while off work rotation), only a portion of soil intake will take place at the site (i.e., during subsistence gathering activities near the road). This FI_s calculation is equivalent to that illustrated in Section 4.2.2.1 for lead exposure. The off-work fractional intake will be equivalent to the fractional intake applied to the subsistence user scenario.

Workers' Consumption of Subsistence Food—Exposure estimates for non-lead chemicals in subsistence foods in the combined worker and subsistence user scenario will use the same formula and input assumptions as in the subsistence user scenario, but with an adjusted fractional intake to account for the differences in subsistence food intake while off work versus while on a work shift (FI_{fw}):

$$\text{Intake } (\text{mg/kg-day}) = \frac{C_f \times 10^{-3} \times CR_f \times ED \times EF}{BW \times AT} \times FI_{fw}$$

The intakes from all food items will then be summed to obtain an estimated subsistence food intake rate.

The combined worker and subsistence user scenario must reflect the differences of subsistence food consumption during the two-thirds of the time that a person is at work and the one-third of the time they are not at work. For the DMTS risk assessment, it will be assumed that 100 percent of food intake while off work is subsistence food, and that the fractional intake while off work (FI_{w_off}) is the same as in the subsistence user scenario (i.e., $FI_{w\_off} = FI$). It is not possible to quantify the amount of subsistence food consumption that occurs while on a work shift, but it would be expected to be low because all meals are provided in the Red Dog cafeteria. For the DMTS risk assessment, it will be conservatively assumed that 25 percent of food intake while on a work shift is subsistence food. Thus, the fractional intake while on a work shift (FI_{w_on}) is 25 percent of the subsistence user scenario (i.e., $FI_{w\_on} = 0.25FI$). Thus, FI_w will be derived by combining the estimates for FI_{w_off} and FI_{w_on} with the relative amounts of time that a worker spends off work (i.e., 0.33) and on a work shift (i.e., 0.67), respectively:

$$\begin{aligned} FI_w &= (0.33 \times FI_{w\_off}) + (0.67 \times FI_{w\_on}) \\ &= (0.33FI) + (0.67 \times 0.25 \times FI) \\ &= (0.33FI) + (0.17FI) \\ &= 0.5FI \end{aligned}$$

4.3 Toxicity Assessment

In the toxicity assessment, the hazards associated with chemicals of concern at the site are evaluated. For noncarcinogenic chemicals, EPA has developed specific toxicity criteria called RfDs. An RfD is an estimate of the level of daily exposure that is likely to be without appreciable risk of health effects over a lifetime, even in sensitive populations. The RfDs used in this assessment for zinc and cadmium are published in EPA's Integrated Risk Information System and are available online (U.S. EPA 2003a). EPA has not developed an RfD for lead, but rather evaluates lead toxicity in reference to blood lead levels. None of the site CoPCs is classified by EPA as a carcinogen for the exposure routes relevant to this assessment.

4.3.1 Barium

Barium is naturally occurring in the environment, present as both a free metal and as barium salts. The most common forms of barium in the environment are barium sulfate and, to a lesser extent, barium carbonate. The form of barium found in the Red Dog ore and ore concentrates is likely barite, the barium sulfate form. In soils, natural barium concentrations range from 15 to 3,000 mg/kg. Barium occurs naturally in almost all surface water bodies and drinking water supplies. However, concentrations are low because the forms generally found in nature are relatively insoluble, particularly in marine waters, where sulfate levels are high (ATSDR 1992). Barium levels in seawater range from 2,000 to 63,000 μg/L, with an average of 13 μg/L (Bowen

1979). In all surface water, concentrations range from 2,000 to 380,000 μg/L (ATSDR 1992). The World Health Organization (WHO 1990) reported that levels of barium in U.S. drinking water range from 1 to 20 μg/L.

Food is the major source of barium intake for most individuals in the general population (ATSDR 1992). WHO (1990) reported the range of daily dietary intake of barium as 300 to 1,770 μg/day. Some nuts, plants, seaweed, and fish, in particular, naturally have relatively high levels of barium. For example, barium concentrations have been reported in corn as 5 to 150 mg/kg, and in various other vegetables as 7 to 1,500 mg/kg (Connor and Shacklette 1975). Gastrointestinal absorption of barium from food has been estimated to be approximately 6 percent in humans (ICRP 1974).

U.S. EPA (2003a) has established an RfD for barium of 0.07 mg/kg-day. The RfD was derived using a weight of evidence approach and the results of four principal studies: 1) an experimental drinking water study in adult people (Wones et al. 1990), 2) an epidemiological study of adults by Brenniman and Levy (1984) of barium in drinking water, 3) a subchronic study of barium in the diet in rats (NTP 1994), and 4) a chronic study of barium in the diet in rats (NTP 1994). U.S. EPA (2003a) derived a NOAEL from both human studies (Wones et al. 1990; Brenniman and Levy 1984) for the critical health effect of hypertension of 0.21 mg/kg-day (after adjustment for water intake and body weight in the study population). The NOAEL was divided by an uncertainty factor of 3 to account for inadequate data in children and the lack of developmental toxicity studies. The resulting RfD was 0.07 mg/kg-day (i.e., 0.21/3=0.07).

In the rodent studies (NTP 1994), rats were fed barium chloride in the diet for either 13 weeks (in the subchronic study) or 2 years (or lifetime, in the chronic study). Barium chloride is a much more soluble (and thus, more bioavailable) form of the metal than the sulfate or carbonate forms that are present in environmental settings. The investigators examined all major organ systems, including the kidney, neurological, and cardiovascular systems. A significant increase in kidney weight was reported in female rats in both the subchronic (at 115 mg/kg-day) and chronic (at 75 mg/kg-day) studies, although no functional or histological changes were noted in the kidneys in the chronic study. No effects were reported at concentrations as high as 45 mg/kg-day in the chronic study. U.S. EPA (2003a) interpreted the results of the NTP (1994) rat studies as showing a lowest-observed-adverse-effect level (LOAEL) for increased kidney weight in females of 75 mg/kg-day and a NOAEL of 45 mg/kg-day.

U.S. EPA (2003a) does not consider barium likely to cause cancer in humans based on the results of studies in rats and mice. The RfD of 0.07 mg/kg-day derived by U.S. EPA (2003a) will be used in the DMTS risk assessment.

4.3.2 Cadmium

Cadmium is a naturally occurring ubiquitous metal constituting 10 to 100 μg/kg of the earth's crust (ATSDR 1999b). The average soil cadmium concentration in the United States is about 250 μg/kg, but varies greatly geographically. Cadmium is not usually present in the environment as a pure metal, but as complex oxides, sulfides, and carbonates. The cadmium that is present in the Red Dog ore and ore concentrates is primarily in the form of sulfides. Cadmium concentrations in most drinking water supplies in the United States are less than

1 μg/L, well below the drinking water standard of 50 μg/L. Levels in drinking water, however, may vary greatly depending on local conditions.

Food and cigarette smoke are the biggest sources of cadmium exposure for people in the general population (ATSDR 1999b). Average cadmium levels in U.S. foods range from 2 to 40 μg/kg, with the lowest levels found in fruits and beverages, and highest levels in leafy vegetables and potatoes. An average American consumes approximately 30 μg of cadmium in their diet each day, but only about 3 to 10 percent (approximately 1 to 3 μg) is absorbed and enters the body. Smokers take in an additional 1–3 μg of cadmium into their body from each pack of cigarettes smoked. One cigarette contains from 1 to 2 μg of cadmium, and 40–60 percent of the cadmium in the inhaled smoke can pass through the lungs into the body. Thus, smokers have about double the cadmium intake of nonsmokers. It is important to consider environmental cadmium exposures in the context of background cadmium exposures from the diet and smoking.

EPA has established two oral RfDs for cadmium: one for food and one for water (U.S. EPA 2003a). The drinking water cadmium RfD is 0.0005 mg/kg-day and the dietary RfD is 0.001 mg/kg-day. Both RfDs are based on human studies indicating that 200 μg per gram kidney is the highest level not associated with proteinuria, or the appearance of protein in the urine (an indicator of kidney dysfunction). A pharmacokinetic model was used to predict the cadmium dose associated with this kidney cadmium level. Assuming 2.5 percent absorption of cadmium from food and 5 percent from water, the pharmacokinetic model predicts that the NOAEL for chronic cadmium exposure is 0.005 and 0.01 mg/kg-day from water and food, respectively. An uncertainty factor of 10 was applied to each of these NOAELs as an additional protection for sensitive individuals who may not have been represented in the studies on which the RfDs were based.

EPA does not consider cadmium to be carcinogenic when exposure occurs by the oral route. Seven studies in rats and mice have shown no evidence of carcinogenic response after cadmium was given orally to the animals. There is no evidence in humans that cadmium is carcinogenic after oral exposure (ATSDR 1999b; U.S. EPA 2003a). EPA does consider cadmium to be carcinogenic when it is inhaled, and it has an inhalation unit risk of 1.8×10^{-3} $(\mu g/m^3)^{-1}$. The EPA unit risk for cadmium is based on a study in which lung cancer was elevated among cadmium smelter workers (Thun et al. 1985), who were exposed to cadmium oxide dust and fume (U.S. EPA 1999c, 2003a). Urinary cadmium data available for a subset of this population suggested high cadmium exposure. However, other risk factors in the study population limit determination of a causal relationship with cadmium (i.e., smoking and prior inhalation exposure to arsenic during prior operation of the facility as an arsenic smelter). Additional studies identified by EPA, in which cadmium exposure was linked to lung cancer, were also noted as having uncertainties related to confounding by concurrent smoking and or arsenic exposure, or were of small populations. EPA is conducting an investigation of the data in Thun et al. (1985) to evaluate the degree to which smoking or prior arsenic exposures partially accounted for the observed lung cancer mortality in this population.[8] In commenting on the Thun et al. (1985) study, EPA noted that:

[8] http://cfpub.epa.gov/ncea/cfm/recordisplay.cfm?deid=22435

> "As the SMRs [Standardized Mortality Ratios] observed were low and there is a lack of clear cut evidence of a causal relationship of the cadmium exposure only, this study is considered to supply limited evidence of human carcinogenicity."

Nevertheless, because there was also a significant and dose-related increase in lung cancer in inhalation investigations with Wistar rats exposed to cadmium chloride aerosol, the database was considered sufficient to constitute evidence of carcinogenicity (Takenaka et al. 1983). The occupational database from Thun et al. (1985) was used as a basis for the unit risk to avoid uncertainties related to converting data from animal studies to predict human risks and because the cadmium oxide exposure was thought to be more representative of environmental exposures (U.S. EPA 1999c, 2003a).

Thus, lung cancer has been observed in workers exposed to high concentrations of cadmium oxide dust and fumes together with other exposures in the smelter setting and in animals exposed to cadmium chloride aerosol. However, neither of these settings is representative of the potential exposures relevant to the risk assessment.

In addition, the pathway screening (described in the CSM section), which compares soil RBCs based on the oral RfD with the RBC based on the EPA unit risk for inhalation of cadmium in dust generated from soil, shows the relative lack of importance of the inhalation pathway. This screening indicated that the soil RBC based on inhalation (with a 10^{-6} risk level) was 1,405 mg/kg. In contrast, the soil RBC based on soil ingestion (with a hazard index of 0.1) was 3.9 mg/kg. This indicates that the relative risk for the inhalation pathway is nearly 400 times lower than that for soil ingestion (i.e., 1,405 mg/kg/3.9 = 380). Also, the RBC derived to be protective of inhalation risk is 3.6 times higher than the maximum site exposure concentration in soil of 388 mg/kg. Thus, if there were any risks related to the inhalation of cadmium in dust generated from soil, they would be expected to be lower than acceptable levels (i.e., less than 1×10^{-6}).

4.3.3 Lead

Lead is a naturally occurring metal found in the Earth's crust, typically at concentrations ranging from 10 to 30 mg/kg (ATSDR 1999a). It is ubiquitous in the environment, both from naturally occurring sources and from its widespread history of use in gasoline, paints, solder for water pipes, and other products. Lead concentrations in Alaska soils range from less than 4 to 310 mg/kg, with a mean of 14 mg/kg (Dragun and Chiasson 1991). Lead levels in U.S. surface and groundwater typically range from 5 to 30 μg/L, with a mean of 3.9 μg/L (ATSDR 1999a). Air lead concentrations in the United States range from 0.001 to 0.005 μg/m^3 in rural settings to an average of 0.04 μg/m^3 in urban settings. The NAAQS for lead is 1.5 μg/m^3. Lead is also present in foods. As reported in ATSDR (1999a), data from the FDA's 1990–1991 Total Diet Study indicate that typical dietary intake of lead at that time ranged from 1.8 to 4.2 μg/day.

The concentration of lead in the blood, typically expressed in micrograms of lead per deciliter whole blood (μg/dL), is generally considered the best measurement for assessing lead exposure and the potential for health effects (CDC 1997). Lead is assessed for its impact on overall blood lead levels. Blood lead levels have declined dramatically in recent decades. For example, in

young children aged 1–5 years who generally have the highest blood lead levels, the national average dropped from 15 μg/dL in 1976–1980 to 2.7 μg/dL when measured between 1991–1994 (Pirkle et al. 1994; Goodman 1997).

The critical (i.e., most sensitive) effects of lead at the lowest blood lead levels associated with environmental exposure are subtle neurobehavioral effects in young children (ATSDR 1999a), although the actual significance of these effects at lower blood lead levels is controversial because these effects become indistinguishable from other factors related to socioeconomic influences (e.g., nutrition and education). Subclinical effects on the blood-forming system are a secondary issue. A blood lead level of 10 μg/dL for children set by the Centers for Disease Control and Prevention (CDC) is the initial level generally used in screening for lead exposure (CDC 1997). For blood lead levels at and above 10 μg/dL, CDC recommends progressively more aggressive follow-up depending on the amount of blood lead elevation (CDC 1997). Initial follow-up begins with education and re-measurement to verify that the blood lead level is 10 μg/dL or greater. Clinical evaluation, environmental investigation, and lead hazard control are not triggered until blood lead levels reach 20 μg/dL or higher. EPA has also identified 10 μg/dL for management of lead exposure in young children and the developing fetus in pregnant women. EPA's risk management guidelines for lead in soil specify limiting exposure "such that a typical (or hypothetical) child or group of similarly exposed children would have an estimated risk of no more than 5 percent of exceeding 10 μg/dL (Laws 1994).

For adults, peripheral neuropathy (i.e., foot drop and wrist drop) or kidney effects have been associated with excessive occupational exposure to lead. A more sensitive effect at lower blood lead levels may be hypertension, based on some epidemiological studies correlating blood pressure with blood lead levels (ATSDR 1999a). Even in cases where a significant association was reported, however, the increase in blood pressure was very slight (Schwartz 1995). EPA considers the fetus of pregnant women to be the sensitive subgroup for lead exposure to adults. Prenatal exposure is likely to be less than exposure in young children, because exposure to the fetus is mediated by the mother, who has a lower lead absorption rate and would ingest less soil or paint chips. Exposure to the fetus also appears less critical for later mental development than at the age of two, according to a statistical evaluation of a number of studies (Pocock et al. 1994). This evaluation reported that no effect of lead exposure was found prior to birth on later mental development in the absence of additional exposure during early childhood.

Federal workplace guidelines for lead exposure differ from EPA guidelines. A blood lead level of 30 μg/dL to protect the reproductive health of workers is recommended by OSHA for workers exposed to lead in the workplace as a part of their employment. Blood lead monitoring is also a part of the requirements. Requirements of the Mine Safety and Health Administration are similar. These regulations, not EPA risk assessment guidelines, would be applicable to workers at the mine.

EPA has classified lead salts as probable human carcinogens (Class B2) based on evidence in animal studies (U.S. EPA 2003a). Although administration of relatively high doses of lead phosphates and acetates to rodents resulted primarily in kidney tumors, a clear relationship was lacking between the lead dose and the incidence of tumors (U.S. EPA 2003a). U.S. EPA (2003a) considers data from the studies to be inadequate to determine the carcinogenic potential of lead. All available studies lacked quantitative exposure data and information on the

contribution from smoking or exposures to other potentially carcinogenic chemicals. EPA recommends against quantitatively evaluating lead as a carcinogen. EPA concluded that lead should be assessed for potential non-carcinogenic effects.

4.3.4 Zinc

Zinc is a naturally occurring ubiquitous metal constituting 0.004 percent (by weight) of the earth's crust (Eisler 2000). The zinc that is present in the Red Dog ore and ore concentrates is primarily in the form of sulfides, originating from the mineral sphalerite. High concentrations of zinc are found in seafood, meats, whole grains, dairy products, nuts, and legumes. It is a nutritionally required trace element in humans and other species. The recommended daily allowance (RDA) for zinc, or the minimum amount required in a person's diet to maintain proper health, is 11 mg/day for adult males, 8 mg/day for adult females, and 12 mg/day for pregnant women. Approximately 20–30 percent of an oral dose of zinc is absorbed through the gastrointestinal tract. Absorption is generally mediated by a homeostatic mechanism over a range of concentrations, and is influenced by various hormones, such as prostaglandins E2 and F2. As a result, exposure to zinc concentrations resulting in toxicity is relatively uncommon and requires very high doses (Goyer 1996). Absorption of zinc can be impeded by a number of organic and inorganic compounds, such as lignin, hemicellulose, cadmium, copper, calcium, and ferrous iron (U.S. EPA 2003a).

EPA has established an oral RfD of 0.3 mg/kg-day based on a clinical study examining copper and iron status in females receiving zinc supplements (U.S. EPA 2003a; Yadrick et al. 1989). The 10-week study of 18 healthy women given zinc gluconate supplements twice daily (50 mg zinc/day) resulted in a decrease of erythrocyte superoxide dismutase (ESOD) activity (Yadrick et al. 1989). ESOD activity, considered a sensitive indicator of copper status, declined to 53 percent of pre-trial levels ($p < 0.01$) by the end of the study. There were also significant reductions in serum ferritin and hematocrit. The RfD was calculated from a total intake, using the LOAEL of 50 mg/day and an assumed dietary intake of 9.72 mg/day, with an average body weight of 60 kg. An uncertainty factor of 3 was applied based on a minimal LOAEL from a moderate-duration study of the most sensitive humans, and consideration of a substance that is an essential dietary nutrient. It is noteworthy that an intake level equivalent to the RfD for pregnant women of 18 mg/day (i.e., 0.03 mg/kg-day × 60 kg body weight) is only slightly higher than the RDA of 12 mg/kg for pregnant women. This highlights the conservative nature of the RfD and ensures that a risk assessment using this RfD will be highly protective of public health.

No positive correlation has yet been established between zinc exposures and increased cancer rates. As a result, zinc has been classified in Group D under EPA's weight of evidence for human carcinogenicity. Group D compounds are not classifiable as to human carcinogenicity, and subsequently, no CSFs have been established. Because of the role of zinc as an essential nutrient and the widespread exposure to this element, carcinogenicity in humans is doubtful at environmental exposure levels. Experimental animals have been given 100 times their dietary requirements without apparent effects and oral administration to animals has not produced carcinogenic effects (Goyer 1996).

4.4 Risk Characterization

In risk characterization, quantitative exposure estimates and toxicity factors are combined to calculate numerical estimates of potential health risk. In this section, potential noncancer health risks will be estimated assuming long-term exposure to contaminants detected in site media. (There are no site CoPCs classified as carcinogens by EPA for the exposure routes relevant to this assessment.) The risk characterization methods described in DEC and EPA guidance will be applied to calculate potential RME and typical excess lifetime cancer risks for carcinogens and hazard indices for contaminants with noncancer health effects. These methods and the results of the risk characterization are described briefly here, and will be fully discussed in the HHRA when completed.

4.4.1 Non-Carcinogenic Risk

With the exception of lead, risks associated with exposure to noncarcinogenic chemicals are evaluated by comparing estimated intake levels with RfDs, and calculating a hazard quotient:

$$\text{Hazard Quotient} = \frac{\text{Intake}}{\text{RfD}}$$

A hazard quotient less than 1 implies that exposure is below the level that is expected to result in a significant health risk. A hazard quotient greater than 1 does not necessarily mean that an effect would occur, rather that exposure may exceed a general level of concern for potential health effects in sensitive populations.

A hazard quotient will be calculated for each CoPC (other than lead) for each of the primary exposure pathways identified in the refined conceptual site model. For each receptor, a hazard index that represents cumulative risk for a CoPC from all exposure pathways will be derived by summing the individual pathway hazard quotients for that CoPC. Cumulative effects from multiple CoPCs will be evaluated where appropriate (i.e., for CoPCs that have similar toxicologic endpoints), by summing the CoPC-specific hazard indices for each receptor.

Risks associated with exposure to lead in each receptor population are expressed in two ways:

1. The predicted geometric mean of blood lead is compared to the EPA target blood lead level of 10 μg/dL
2. The predicted probability of exceeding the target blood lead level is compared to the target probability of 5 percent.

Values less than the target levels imply that exposure is below the level that is expected to result in a significant health risk. Values greater than the target levels do not necessarily mean that an effect would occur, rather that exposure may exceed a general level of concern for potential health effects in sensitive populations.

4.4.2 Carcinogenic Risk

There are no site CoPCs classified as carcinogens by EPA for the exposure routes relevant to this assessment.

4.4.3 Uncertainty Assessment

Risk assessment is subject to a number of uncertainties. General sources of uncertainty include the site characterization (adequacy of the sampling plan and quality of the analytical data), the exposure assumptions, estimation of chemical toxicity, background concentrations, and the present state of the science involved. Uncertainties in the assessment will be discussed and, where possible, addressed by conducting additional analyses.

5 Ecological Risk Assessment

The purpose of the baseline ERA is to determine if exposures to CoPCs in terrestrial and aquatic environments along the DMTS road corridor result in adverse effects to ecological receptors that utilize the site. The following sections propose a methodology for quantifying and interpreting ecological risks. In Section 5.1, the problem formulation and the results of the ecological screening assessment, presented above in Sections 3.5 and 3.6, as well as knowledge of site ecology, are used to determine the scope and focus of the ERA. Refinement of CoPCs, identification of complete exposure pathways, and selection of assessment endpoints, measurement endpoints, and representative ecological receptors are discussed in this section. Sections 5.2 and 5.3 outline approaches for evaluating risks to communities of lower trophic-level organisms that may be exposed to CoPCs at the site. Methods for assessing risks to avian and mammalian receptors are proposed in Section 5.5: deterministic and probabilistic approaches to modeling dietary exposures to CoPCs are presented, followed by discussions of TRVs and risk calculations, in which estimated dietary exposures are compared to TRVs to evaluate the levels of risk posed by CoPCs. The approach for characterizing the ecological significance of risk estimates and identifying uncertainties that may affect risk estimates is briefly described in Section 5.6.

5.1 Problem Formulation

The problem formulation for the ERA draws upon the results of the screening assessment and the site-specific knowledge acquired through Phase 1 sampling to refine the list of CoPCs and the preliminary conceptual model presented in Section 2.4, including complete exposure pathways, ecological receptors, and assessment and measurement endpoints: complete exposure pathways and relevant receptors are integrated into a refined CSM at the end of this section.

5.1.1 Refinement of CoPCs

CoPCs for the assessment of risk to lower trophic-level organisms were identified for each environment and medium through a tiered screening process that compared chemical concentrations to ecological benchmarks and reference data, as described in Section 3.6.2. Chemicals that failed all screening tiers were retained for further risk analysis. Table 3-40 summarizes by environment the CoPCs that were retained for the ERA. In Section 3.5.6, CoPCs for wildlife were identified by using screening-level food-web models to calculate dietary exposure to CoPCs and then comparing exposures to no-effect level TRVs for those chemicals. These screening results are summarized in Section 3.6.3.

5.1.2 Complete Exposure Pathways

Complete exposure pathways exist for lower trophic-level organisms and wildlife associated with several environments at the site, via direct contact, uptake, or ingestion of soil or sediment and ingestion of food. Complete exposure pathways in each environment are discussed below.

5.1.2.1 Terrestrial

Compared to the primary exposure pathways (direct contact, uptake, and ingestion), inhalation of soil particles in the terrestrial environment is considered a secondary exposure pathway that may represent minor exposures for birds and mammals, as discussed in Section 2.4.2. Therefore, in the tundra environment, the baseline assessment will evaluate risk to terrestrial plants from uptake of chemicals from soil and fugitive dust deposition, risk to soil fauna from direct contact with and uptake or ingestion of chemicals in soil, and risk to terrestrial birds and mammals from ingestion of chemicals in food and soil, or surface water consumed from streams or tundra ponds.

5.1.2.2 Streams

In streams, the baseline assessment will evaluate risk to aquatic or wetland plants and aquatic invertebrates from direct contact with or uptake/ingestion of chemicals dissolved in surface water, uptake/ingestion of chemicals in sediment, and ingestion of chemicals in food (for aquatic invertebrates). No chemicals were retained as CoPCs on the basis of AWQC and reference area exceedances in stream water, the primary exposure medium for fish. Ott and Morris (2004) concluded that cadmium, lead, selenium, and zinc concentrations in fish from Aufeis Creek and the Omikviorok River are low compared to levels in fish from streams near the mine, and that there is no indication that chemical concentrations increase in fish captured downstream of the DMTS road relative to upstream reference locations. In Anxiety Ridge Creek, the study found no indication that selenium or zinc concentrations increase in fish collected downstream of the DMTS road, but that the cadmium and lead concentrations in fish tend to be higher downstream of the road. However, cadmium and lead concentrations in Anxiety Ridge Creek do not appear to be increasing over time and are comparable to levels in fish from other streams in active or proposed mining areas in Alaska (Ott and Morris 2004). The report recommends discontinuing juvenile Dolly Varden sampling in Aufeis Creek and the Omikviorok River and focusing the monitoring program on streams near the mine Ott and Morris 2004). Therefore, based on the results of the ecological screening for stream water and biomonitoring results in streams that cross the DMTS road, risks to fish appear to be negligible, and further assessment of these receptors in the baseline assessment is not warranted.

The quantitative risk assessment for aquatic birds and mammals that forage in streams will estimate exposures to chemicals from food ingestion (i.e., aquatic plants and invertebrates) and the incidental ingestion of sediment. Screening exposure models indicate that the likelihood of adverse effects to avian and mammalian piscivores foraging in streams and creeks is low, as hazard quotients were typically much lower than 1.0, and chemical concentrations in fish appear to be similar to concentrations at reference locations. Therefore, further evaluation of risk to piscivores foraging in these habitats is not required. Screening models indicate that there is a potential for adverse effects to avian invertivores from exposure to lead and selenium in prey or sediment, and these risks will be quantified in the baseline assessment. Currently there is insufficient data to evaluate risk to herbivorous birds and mammals that may use streams and creeks, and risk to these receptors will be evaluated in the baseline assessment.

5.1.2.3 Tundra Ponds

Risks to aquatic or wetland plants and herbivorous birds and mammals will also be assessed in the tundra pond environment, as there is insufficient data at present to screen out these pathways. Screening results indicate that there could be adverse effects to benthic invertebrates and invertivorous birds that inhabit or forage at tundra ponds. Exposure pathways to these receptors, however, may be incomplete in the tundra ponds based on results of the Phase 1 investigation. The substrate of these ponds consists of dense vegetation mats that appear to represent sub-optimal habitat for invertebrates. Preliminary sampling conducted during the Phase 1 field event found no benthic invertebrates in the tundra ponds. However, aquatic invertebrates are known to utilize tundra pond habitats from studies conducted elsewhere in Alaska (USFWS 1984), and therefore, because the absence of complete exposure pathways cannot be conclusively determined, risk to aquatic invertebrates will be assessed in the baseline assessment. An FWS report on the ecology of tundra ponds of the Arctic Coastal Plain (USFWS 1984) stated that when feeding, "wading shorebirds utilize the tundra itself and exposed sediments of temporary wetlands rather than the ponds or lakes." Thus, ingestion of tundra pond invertebrates appears to be a secondary exposure pathway to invertivores that may feed in tundra pond habitats at the site, such as the common snipe, but may still represent a complete exposure route. Therefore, risk to freshwater avian invertivores that feed on aquatic and terrestrial invertebrates around the fringes of tundra ponds will be evaluated in the risk assessment. These organisms may be exposed to CoPCs primarily through incidental ingestion of sediment and ingestion of food.

Based on observations from the Phase 1 sampling event, complete exposure pathways to fish or subsequently to piscivorous wildlife do not exist in the tundra pond environment. The tundra ponds observed at the site and reference area in Phase 1 were hydrologically disconnected from surface water inputs from streams and creeks and tended to be shallow areas of flooded tundra that may contract or disappear during dry periods (Photographs 4 and 5). As such, these ponds are unlikely to support permanent fish populations, and no fish were observed in the ponds sampled in Phase 1. Therefore, pathways to fish and piscivorous wildlife are considered to be incomplete in tundra ponds, and these receptors will not be assessed in this environment in the baseline risk assessment.

5.1.2.4 Coastal Lagoons

Pathways to aquatic and wetland plants, aquatic invertebrates, and herbivorous and invertivorous wildlife exist in coastal lagoons, and risks to these receptors will be assessed in the baseline assessment. No chemicals were retained as CoPCs on the basis of AWQC and reference area exceedances in surface water, the primary exposure medium for fish that may inhabit these lagoons; therefore, pathways to fish are considered incomplete and will not be assessed further. Risks to piscivorous birds and mammals that may feed on fish in coastal lagoons will not be assessed quantitatively in the baseline assessment, because the results of the food web models for freshwater piscivores showed no potential for adverse effects, and those results are considered protective of piscivores that forage in coastal lagoons, due to the lower likelihood of chemical accumulation in marine fish relative to freshwater fish.

5.1.2.5 Coastal Marine

Because no chemicals in coastal marine surface water or sediment exceeded reference area concentrations and relevant ecological screening values in at least 10 percent of samples analyzed, further evaluation of this habitat in the baseline assessment is not warranted.

5.1.3 Selection of Assessment Endpoints

Assessment endpoints are components of the ecosystem that represent important environmental values and that may be susceptible to adverse effects from exposure to chemicals in fugitive dust. Preliminary assessment endpoints for the ERA were identified in Section 2.4.6. The preliminary assessment endpoints in each environment were refined based on the results of the ecological screening and site-specific observations from the Phase 1 sampling event. There are eight assessment endpoints in the terrestrial tundra environment:

- Structure and function of:
  - Terrestrial plant communities
  - Tundra soil fauna communities.
- Survival, growth, and reproduction of terrestrial avian:
  - Herbivore populations
  - Invertivore populations
  - Carnivore populations.
- Survival, growth, and reproduction of terrestrial mammalian:
  - Herbivore populations
  - Invertivore populations
  - Carnivore populations.

There are five assessment endpoints in the stream environment:

- Structure and function of:
  - Stream aquatic and wetland plant communities
  - Stream aquatic invertebrate communities.
  - Survival, growth, and reproduction of stream avian:
  - Herbivore populations
  - Invertivore populations.

- Survival, growth, and reproduction of stream mammalian:
  - Herbivore populations.

There are five assessment endpoints in the tundra pond environment:

- Structure and function of:
  - Tundra pond aquatic and wetland plant communities
  - Tundra pond aquatic invertebrate communities.
- Survival, growth, and reproduction of tundra pond avian:
  - Herbivore populations
  - Invertivore populations.
- Survival, growth, and reproduction of tundra pond mammalian:
  - Herbivore populations.

There are four assessment endpoints in the coastal lagoon environment:

- Structure and function of:
  - Coastal lagoon aquatic and wetland plant communities
  - Coastal lagoon aquatic invertebrate communities.
- Survival, growth, and reproduction of coastal lagoon avian:
  - Herbivore populations
  - Invertivore populations.

Because the screening results indicate that a baseline risk assessment is not warranted in the coastal marine environment, no assessment endpoints are retained for that environment. Table 5-1 summarizes the assessment endpoints for the ERA.

5.1.4 Selection of Measurement Endpoints

Measurement endpoints provide the actual parameters used to evaluate attainment of each assessment endpoint. The refined list of measurement endpoints for the DMTS risk assessment is presented in Table 5-1.

The measurement endpoints used to evaluate the impacts to assessment endpoints, such as the structure and function of plant and invertebrate communities, will focus on evaluation of community-level parameters for these endpoints, as described in greater detail in following

sections. For assessment endpoints such as the survival, growth, and reproduction of various bird and mammal populations, the measurement endpoints are the range of modeled dietary exposures of each representative receptor to CoPCs (based on measured CoPC concentrations in food, soil, sediment, and surface water) as compared to TRVs derived from the literature.

5.1.5 Ecological Receptors

The following sections describe the ecological receptors selected to represent functional groups, such as terrestrial mammalian herbivores or freshwater aquatic avian invertivores, in the quantitative wildlife exposure assessment. Section 2.4.6 provides a brief discussion of the methods used to choose appropriate wildlife receptors. Thirteen wildlife receptors will be evaluated in the risk assessment:

- Willow ptarmigan
- Tundra vole
- Caribou
- Moose
- Lapland longspur
- Tundra shrew
- Snowy owl
- Arctic fox
- Green-winged teal
- Muskrat
- Common snipe
- Brant
- Black-bellied plover.

5.1.5.1 Terrestrial Receptors

In terrestrial portions of the site, CoPCs have been identified in tundra soil, and therefore risk of adverse effects to terrestrial plants and soil invertebrates from tundra soil exposure will be assessed. Risk of adverse ecological effects to birds and mammals that may feed on plants at the site will be evaluated using food-web models to estimate total dietary exposure to CoPCs. The willow ptarmigan (*Lagopus lagopus*), tundra vole (*Microtus oeconomus*), barren-ground caribou (*Rangifer arcticus granti*), and moose (*Alces alces*) have been selected as receptors representing avian and mammalian herbivores in the food-web model. These four species are

known to occur at the site (DEC et al. 2002) and may be exposed to CoPCs in surface water, soil, and their diet.

The willow ptarmigan is a year-round resident of tussock and shrub tundra in the vicinity of the DMTS road. It is often associated with shrubby willow and birch habitats and eats predominantly willow throughout the year, including the buds, leaves, twigs, and catkins (Hannon et al. 1998). The willow ptarmigan is fairly common in Cape Krusenstern National Monument and Noatak National Preserve and is known to nest in these areas (Schroeder 1996). In 1981–1982 baseline studies, willow ptarmigan was observed in Dryas-dwarf shrub tundra, riparian tall and low shrub, tussock-shrub tundra, and sedge-grass tundra/wet meadow environments (Dames & Moore 1983a). Residents of Kivalina and Noatak harvest ptarmigan and ptarmigan eggs for subsistence use (Sundet 2002a,b, pers. comm.).

The tundra vole inhabits wet meadows, marshes, and other moist areas around the site, where it feeds on grasses, sedges, and other vegetation (Bee and Hall 1956). During 1981–1982 baseline studies, the tundra vole was the only species of small mammal captured in snap and pit fall traps; it was trapped in dwarf shrub tundra habitat near the runway site at the mine (Dames & Moore 1983a). The tundra vole is a default indicator species chosen by DEC for ERAs conducted in the northwest ecoregion (DEC, no date).

The barren-ground caribou occurs seasonally in the vicinity of the DMTS road and the port. The largest numbers arrive during the fall migration, when caribou of the Western Arctic caribou herd cross the DMTS road on their way to winter ranges in river drainages south of the site (Hemming 1987, 1988, 1989, 1990, 1991; Pollard 1994a,b). In most years, a small percentage of the migrants may remain near the site throughout the winter. Fewer caribou are observed at the site during the spring and summer than during the fall migration. The barren-ground caribou browses on a wide range of lichens, mosses, grasses, sedges, forbs, and shrubs during the growing season and utilizes lichens heavily in the winter (Bee and Hall 1956; Bergerud 1972; Holleman et al. 1979). Residents of Kivalina and Noatak harvest caribou throughout the year (Sundet 2002a,b, pers. comm.).

The moose is a large resident herbivore that forages in a variety of habitats at the site, from alpine shrub areas near the mine to riparian habitats near the coast (Dames & Moore 1983a). The moose is primarily a browser, particularly during winter, when it feeds on twigs, bark, and senescent leaves of willows, birch, and other woody plants (Peek 1974; Risenhoover 1989; DFG 2003e). During the growing season, moose may consume grasses, sedges, horsetails, forbs, and emergent and submerged aquatic vegetation, in addition to browse species (Peek 1974; Risenhoover 1989; DFG 2003e). Dames & Moore (1983a) reported numerous moose sightings during their 1981–1982 baseline studies, but the authors suggested that the total moose population at the site was relatively small and observed that moose were absent from large tracts of suitable habitat. Residents of Kivalina and Noatak hunt moose in the region (Sundet 2002a,b, pers. comm.).

Adverse ecological effects can also occur in higher trophic-level species, both through direct exposure to CoPCs in environmental media and consumption of prey containing these CoPCs. Therefore, risk of adverse ecological effects to avian invertivores, mammalian invertivores, avian carnivores, and mammalian carnivores that may feed at the site will be evaluated by

modeling total dietary exposure to CoPCs for the Lapland longspur (*Calcarius lapponicus*), the tundra shrew (*Sorex arcticus tundrensis*), the snowy owl (*Nyctea scandiaca*), and the arctic fox (*Alopex lagopus*), respectively. These species may be exposed to CoPCs in soil, surface water, and their diet (Table 2-5).

The Lapland longspur migrates annually from wintering grounds in temperate North America to breeding grounds on the arctic tundra (Hussell and Montgomerie 2002). This species arrives at the port site in May and is among the most prevalent birds in tussock-shrub tundra habitat; it also occurs in sedge-grass wet meadow, riparian tall and low shrub, and coastal tall grass habitats (Dames & Moore 1983a). The Lapland longspur is abundant in Cape Krusenstern National Monument and Noatak National Preserve and is known to nest in both parks (Schroeder 1996). Its summer diet consists mainly of arthropod larvae and adults, but it relies on seeds and plant material during the winter (Hussell and Montgomerie 2002). The Lapland longspur is the default indicator species chosen by DEC to represent terrestrial avian invertivores for risk assessments conducted in the northwest ecoregion (DEC, no date).

The tundra shrew (also known as the arctic shrew) is found across northern North America (University of Michigan 2002) and is expected to occur at the study area, although no shrews were captured in snap and pit fall traps during the 1981–1982 baseline study (Dames & Moore 1983a). Well-drained areas bordering on wetlands, streams, or wet tundra are typical habitats for this species (University of Michigan 2002; YDRR 2002). The tundra shrew eats a diverse diet of invertebrates such as beetles, worms, spiders, slugs, snails, and insect larvae (University of Michigan 2002). It is the default indicator species chosen by DEC to represent terrestrial mammalian invertivores for risk assessments conducted in the northwest ecoregion (DEC, no date).

The snowy owl occurs in ocean beach, tussock-shrub tundra, and sedge-grass wet meadow habitats in the study area during the breeding season (Dames & Moore 1983b). It nests on open, elevated sites such as hummocks and boulders that overlook the surrounding tundra, where it hunts small mammals, such as rodents and hares, as well as small to medium-sized songbirds and waterfowl (Parmelee 1992). The snowy owl may remain in its breeding range throughout the year or may migrate south in the winter (Parmelee 1992). Residents of Kivalina and Noatak harvest at least three species of owls, including snowy owls (Sundet 2002a,b, pers. comm.).

The arctic fox is a permanent resident of the tundra in the vicinity of the DMTS road (Dames & Moore 1983a). It preys on small mammals and birds but will also eat eggs, carrion, berries, and plants when available (Chesemore 1975). Foxes were among the small mammals that Kivalina residents mentioned during the subsistence discussion on June 17, 2002 (Sundet 2002a, pers. comm.).

5.1.5.2 Freshwater Aquatic Receptors

CoPCs have been identified in sediment from streams that cross the DMTS road, as well as tundra ponds located in the DMTS road corridor, and therefore risk of adverse ecological effects to freshwater aquatic and wetland plants and aquatic invertebrates that may contact or take up chemicals from these sediments will be assessed (Table 5-1). Risk of adverse ecological effects to birds and mammals that may consume freshwater plants at the site will be assessed using the

green-winged teal (*Anas crecca*) and the muskrat (*Ondatra zibethicus*) as receptors representing freshwater herbivores. The teal and the muskrat may be exposed to CoPCs in surface water, sediment, and their diet (Table 2-5).

The green-winged teal is the smallest North American dabbling duck and an opportunistic consumer of a broad range of seeds and other plant material, aquatic insects, molluscs, and crustaceans (Johnson 1995). It typically feeds in shallow water or on mudflats (Johnson 1995) and was observed in marine (coastal lagoons), lacustrine (ponds), and fluviatile (rivers and streams) waters from May to September during the 1981–82 baseline studies (Dames & Moore 1983a). The green-winged teal is a common nesting bird in Cape Krusenstern National Monument and Noatak National Preserve (Schroeder 1996). Residents of Kivalina and Noatak harvest ducks for food and feathers and collect duck eggs as well (Sundet 2002a,b, pers. comm.). The green-winged teal is the default indicator species chosen by DEC to represent freshwater semi-aquatic avian herbivores for risk assessments conducted in the northwest ecoregion (DEC, no date).

The muskrat, a large, herbivorous rodent, occurs across mainland Alaska south of the Brooks Range (DFG 2003a). Muskrats are present in Cape Krusenstern National Monument and Noatak National Preserve (MacDonald and Cook 2002), and one muskrat was observed in sedge-grass marsh habitat around Kavrorak Lagoon during the 1981–82 baseline studies, indicating that this species probably occurs, at least in low numbers, in the vicinity of the port and the DMTS road (Dames & Moore 1983a). The muskrat eats mainly aquatic plants, including cattails, lilies, grasses, and sedges, which it often tows to a feeding platform and may store for winter consumption. It also feeds occasionally on clams, shrimp, frogs, and small fish (DFG 2003a; Whitaker 1997). Residents of Kivalina and Noatak harvest muskrats for meat and pelts (Sundet 2002a,b, pers. comm.). The muskrat is the default indicator species chosen by DEC to represent freshwater semi-aquatic mammalian herbivores for risk assessments conducted in the northwest ecoregion (DEC, no date).

Risk of adverse ecological effects to birds that may feed on freshwater invertebrates at the site will be assessed using the common snipe (*Gallinago gallinago*) as the representative receptor for freshwater avian invertivores. The snipe may be exposed to CoPCs in surface water, sediment, and its diet (Table 2-5).

The common snipe has been observed in riparian tall and low shrub and sedge-grass wet meadow habitats in the study area during the breeding season (Dames & Moore 1983b) and is known to nest in Cape Krusenstern National Monument and Noatak National Preserve (Schroeder 1996). This species uses its long bill to probe the sediments for larval insects, worms, crustaceans, and mollusks (Mueller 1999). The common snipe is the indicator species selected by DEC to represent freshwater semi-aquatic avian invertivores for risk assessments conducted in the northwest ecoregion (DEC, no date).

5.1.5.3 Coastal Lagoon Receptors

CoPCs have been identified in coastal lagoon sediments at the port site, and complete exposure pathways exist to aquatic plants and invertebrates that may contact or take up chemicals from these sediments. Thus, risk of adverse ecological effects to these receptors will be assessed in

lagoons (Table 5-1). Risk of adverse ecological effects to birds that may feed on aquatic plants or invertebrates near the port site will be assessed using the brant (*Branta bernicla*) as the receptor representing marine avian herbivores and the black-bellied plover (*Pluvialis squatarola*) as the receptor representing coastal avian invertivores.

The brant is a small goose that breeds in the arctic, winters from Alaska south to Baja California, and remains near saltwater throughout the year (DFG 2003b; Reed et al. 1998). It occurs in marine and lacustrine waters, wet meadows and marshes, and sedge-grass tundra environments at the site (Dames & Moore 1983a) and is known to nest in Cape Krusenstern National Monument (Schroeder 1996). The brant feeds almost exclusively on plants, predominantly eelgrass, salt marsh plants, and green algae during the winter and arctic grasses and sedges, forbs, and moss during the breeding season (Reed et al. 1998). It forages on exposed vegetation and rooted plants in shallow water but does not dive; at high tide, it feeds on dislodged leaves floating at the surface (Reed et al. 1998; Hebert 2003). Residents of Kivalina and Noatak harvest geese such as the brant for subsistence use (Sundet 2002a,b, pers. comm.). The brant is the default indicator species chosen by DEC to represent marine semi-aquatic avian herbivores for risk assessments conducted in the northwest ecoregion (DEC, no date).

The black-bellied plover is a shorebird that breeds exclusively in the arctic but winters along the coasts of North, Central, and South America (Paulson 1995; Hebert 2003). It nests in shallow scrapes on dry tundra, gravelly plains, or in coastal marshes (DFG 2003d; Paulson 1995; Hebert 2003) and is known to breed in Cape Krusenstern National Monument (Schroeder 1996). The black-bellied plover was observed in tussock-shrub tundra and sedge-grass, wet meadow, and marsh habitats at the site during the 1981–1982 baseline studies (Dames & Moore 1983a). On its breeding grounds, this species eats mainly insects but also polychaetes, bivalves, crustaceans, and berries. The black-bellied plover is the default indicator species chosen by DEC to represent marine semi-aquatic avian invertivores in the northwest ecoregion (DEC, no date).

5.1.6 Refined Conceptual Site Model

Based on the results of the ecological screening and the site-specific knowledge gained during Phase 1 sampling, the CSM for the DMTS risk assessment was revised to include only complete pathways that may result in CoPC exposures at the site. The refined CSM, illustrated in Figure 5-1, distinguishes among aquatic ecosystems, such as freshwater streams and tundra ponds, and coastal lagoons, to show clearly which pathways and receptors are important in each environment. The refined model also provides a more detailed summary of exposure than the preliminary CSM by defining primary and secondary exposures for receptor guilds (e.g., herbivorous mammals) instead of broad receptor categories (e.g., all mammals). Thus, the refined CSM illustrates exposure pathways specific to each receptor guild to be assessed in the ERA. These pathways are described above in Section 5.1.2. Primary exposure routes will be quantified in the ERA, while secondary exposure routes will be addressed qualitatively in the uncertainty analysis.

5.2 Terrestrial Plant and Soil Fauna Assessment

Evaluation of potential adverse effects to terrestrial plant communities will be accomplished by measuring plant community parameters at various quadrat sampling locations across a range of CoPC concentrations at the site. Community survey methods and measurement endpoints will be developed in detail in the field sampling plan to be prepared in advance of the Phase II investigation. Potential measurement endpoints include species abundance and diversity, biomass, and percent vegetative cover. Similar measurements will be taken at a reference location for comparison with results obtained at the site. Concentrations of CoPCs in tundra soil will also be measured concurrent with plant community analyses, to attempt to relate observed effects, if any, to chemical concentrations. Wherever possible, concentrations measured in media will also be compared to effects ranges reported in the scientific literature.

Limited sampling conducted during the Phase I investigation indicated that soil fauna are not very abundant, even at the reference location. Some invertebrates and small nematodes were observed within moss clumps, but few in the soil. Ecological screening benchmarks for soil are typically much lower for plants than for soil fauna (Table 3-19). Therefore, it is anticipated that if there were adverse effects due to the presence of chemicals in tundra habitats, these effects would be apparent in plant communities at concentrations where no effects would be seen on soil fauna. For this reason, it is assumed for purposes of the baseline risk assessment that results of the terrestrial plant community analysis will be protective of potential adverse effects to soil fauna, and no direct measurement of soil fauna is planned.

5.3 Freshwater and Coastal Lagoon Aquatic Life Assessment

Potential effects on aquatic invertebrate communities will be evaluated by measuring community characteristics, such as species abundance and diversity, across a range of CoPC concentrations in freshwater or coastal lagoon habitats. Sampling locations and methods will be developed in detail in the field sampling plan. Concentrations of CoPCs in sediment will also be measured concurrently to attempt to relate observed effects, if any, to chemical concentrations. Wherever possible, concentrations measured in media will also be compared to effects ranges reported in the scientific literature. Preliminary sampling as part of the Phase I investigation indicated that benthic invertebrates were uncommon in ponds, although this could be due to the time of the season when sampling was conducted or the methods used to collect invertebrates. Further evaluation of sampling methods will be presented in the field sampling plan, and alternative measurement approaches, such as conducting *ex situ* toxicity tests, will be discussed.

The anticipated approach for evaluating aquatic plant and wetland plant communities is similar to that described above for terrestrial plant communities. Observations made during the Phase I investigation indicate that in some habitats, particularly creeks and streams, aquatic plants are sparse, likely due to the physical characteristics of the waterbodies (i.e., cobble substrate with little sediment accumulation). Factors such as this will be taken into consideration when designing a field sampling plan, as they may limit the ability to identify a sufficient number of locations where plant community analysis can be performed. Concentrations of CoPCs in environmental media will also be measured concurrent with plant community analyses to

attempt to relate observed effects, if any, to chemical concentrations. Wherever possible, concentrations measured in media will also be compared to effects ranges reported in the scientific literature.

5.4 Avian and Mammalian Receptor Assessment

To assess ecological risks to birds and mammals, food web models will be structured to estimate site-specific daily doses of CoPCs to these receptors. This approach will allow for a direct comparison of exposure rates with measures of toxicity in the risk characterization. The ratio of an exposure estimate to an ecotoxicity value, such as a TRV, is known as a hazard quotient (U.S. EPA 1997a). Deterministic exposure models describe a single representative exposure scenario for a receptor and CoPC combination in a given environment or assessment unit, such as the daily exposure to lead for a willow ptarmigan feeding along the DMTS road, calculated using point estimates for each exposure variable. Exposure variables in food web models include receptor-specific parameters such as body weight; food, water, and sediment or soil ingestion rates; dietary composition; and area-use factor (ratio of the risk assessment area to the area utilized by the receptor), as well as site-specific CoPC concentrations in dietary components and inert media (U.S. EPA 1997a). Hazard quotients developed as single-point exposure and effects comparisons are useful for identifying potential low- or high-risk situations (63 Fed Reg. 26845–26924). U.S. EPA (1999a) recommends using a point-estimate approach as a first step in risk characterization, before considering more complex risk assessment tools such as probabilistic modeling. Therefore, deterministic exposure models will be developed for all wildlife receptors in the baseline risk assessment.

Probabilistic risk assessment refers to the use of probability models to predict the likelihood of various levels of risk in a population or to characterize the uncertainty in risk estimates (U.S. EPA 1999a). By incorporating the full scope of available information, probabilistic risk analysis may provide a more complete risk characterization than a deterministic assessment, particularly at complex sites like the DMTS road corridor (U.S. EPA 1999a). In probabilistic exposure modeling, the food web models are solved probabilistically over the entire distributions of observed CoPC concentrations and projected receptor population distributions to provide estimates of the probability of a given exposure rate over all possible outcomes. In the DMTS fugitive dust ERA, probabilistic exposure models may be developed to supplement the deterministic analysis in cases where hazard quotients based on single-point exposures exceed unity. The following sections describe methods for deterministic and probabilistic exposure modeling for avian and mammalian receptors.

5.4.1 Deterministic Food-Web Exposure Modeling

Deterministic food web models will be developed to estimate daily dietary exposures to CoPCs for birds and mammals that may feed at the site, and risk calculations using the hazard quotient approach will be applied to evaluate risk to these receptors. Because of its size and diversity, the DMTS study area may be divided into logical assessment units for the purposes of the assessment. Assessment units may be determined based on habitat types where receptors forage (e.g., coastal lagoons versus inland creeks), foraging ranges of receptors, or ranges of CoPC

concentrations measured in media across the site (e.g., at different distances from the DMTS road). Separate exposure models may be developed for each assessment unit to identify areas with the highest potential for adverse ecological effects. For example, screening of chemical concentrations in tundra soil (see Section 3.5.1) indicates that for some chemicals, such as antimony, arsenic, copper, and nickel, most or all of the stations where screening values were exceeded lie within the port area, and therefore the likelihood of risk is greatest at this location. For other chemicals, such as lead and zinc, screening value exceedances are more widely distributed across the entire study area, and therefore a more extensive spatial evaluation of risk is warranted for these CoPCs.

Evaluation of the exposure potential for avian and mammalian receptors will be accomplished using simple deterministic food-web exposure models as described in Section 3.5.6. These models provide an estimated total dietary exposure to CoPCs resulting from consumption of food, water, and the incidental ingestion of sediment or soil on a mg/kg body-weight-day basis. For all receptors modeled, the food-web models will use conservative, but ecologically relevant, values for exposure parameters. The receptor-specific exposure parameters that will be used are shown in Table 5-2. In the absence of data on relative gastrointestinal absorption efficiencies, the parameter A_i will be set to a value of 1.0.

Mean measured CoPC concentrations in biota, water, and sediment or soil will be used to calculate dietary exposures. Section 3.2 discusses the surface water, sediment, and soil data that are currently available for use in food web models. Limited plant data are available to model exposures to CoPCs in food for terrestrial herbivores. Existing site data include cadmium, lead, and zinc in willow leaves and lichen (Exponent 2002a, DEC et al. 2002); three (cadmium, lead, and zinc) to eight (cadmium, chromium, cobalt, copper, lead, manganese, nickel, and zinc) chemicals in salmonberries, blackberries, and sourdock (Exponent 2002a, DEC et al. 2002; E&E 2002; DHSS 2001); and three (cadmium, lead, and zinc) to 21 (aluminum, antimony, arsenic, barium, cadmium, chromium, cobalt, copper, iron, lead, manganese, mercury, molybdenum, nickel, selenium, silver, strontium, thallium, tin, vanadium, and zinc) chemicals in moss (Exponent 2002a; Ford and Hasselbach 2001; Hasselbach 2003b, pers. comm.). Chemical data are not currently available to model exposures to CoPCs in food for freshwater and coastal lagoon herbivores, all invertivores, and all carnivores. A Phase 2 sampling program will be designed to fill data gaps in the risk assessment, such as missing prey data or CoPCs, and to supplement existing biota data where spatial coverage is limited; details of the proposed biota sampling will be provided in the Phase 2 field sampling plan. Data needs are summarized below in Section 7. In the exposure assessment, plant and prey data will be utilized to approximate the composition of each receptor's diet, as summarized in Table 5-2.

Risk of adverse ecological effects to bird and mammal populations will be estimated using the hazard quotient approach:

$$HQ = \frac{IR_{chemical}}{TRV}$$

where:

HQ = hazard quotient (unitless)

$IR_{chemical}$ = total ingestion rate of the chemical (mg/kg body weight-day)

TRV = toxicity reference value (mg/kg body weight-day).

The daily dietary exposure to a CoPC will be compared against the LOAEL TRV. The LOAEL is the minimum dose reported to result in a statistically significant adverse effect in the specific type of receptor. Thus, an exposure rate in excess of the LOAEL TRV may result in an adverse effect to an exposed individual. The LOAEL TRV will be used to describe the potential for adverse ecological effects to occur as a result of CoPC exposure.

Spatial patterns determined by the division of the study site into assessment units will be evaluated to determine the extent of toxicity threshold exceedances and their possible effects on receptor populations. For example, depending on the availability of sufficient data, risk to avian and mammalian wildlife foraging in streams may be assessed on a stream-by-stream basis, on the assumption that each stream likely supports discrete populations of most receptors given the spatial segregation of streams in relation to the potential foraging range of individual receptors (e.g., muskrat or snipe foraging at Aufeis Creek would not also be expected to forage at New Heart Creek). In this way, risks can be determined separately for each creek. Similarly, each coastal lagoon likely represents a discrete habitat for species with limited foraging ranges, such as the black-bellied plover.

However, the delineation of discrete spatial units is more difficult in the tundra habitat (with the possible exception of the area defined by the port site boundaries), given the uniformity of the habitat along the DMTS transport corridor and the manner in which habitat use may vary by receptor. For example, voles have a very limited home range, and risk estimates are likely to be very different for a vole with a foraging range immediately adjacent to the DMTS road relative to a vole whose territory is a kilometer or more distant from the road. Consequently, the risk calculated for either of these two individuals does not accurately depict the risk to the vole population as a whole within the study area. At the other end of the spectrum, risk to widely ranging terrestrial receptors, such as caribou, may be integrated over the entire site, and the magnitude of risk will depend on how long an individual caribou forages in a given location. Because of the spatial extent of the terrestrial tundra habitat, additional analytical methods beyond simple deterministic food web models may be required to accurately quantify population-level risks. Therefore, an additional probabilistic approach to ERA for the terrestrial environment is described below.

5.4.2 Probabilistic Food-Web Exposure Modeling

In cases for terrestrial wildlife where exposure assessments using deterministic food-web models result in hazard quotients exceeding 1.0 based on comparison to the LOAEL TRV, probabilistic risk assessment methods may be used to determine the proportion of the population projected to incur an adverse impact, defined as that portion of the total population whose exposure exceeds the LOAEL TRV distribution. Probabilistic food web models used to estimate the rate of CoPC exposure will be structured to estimate site-specific daily doses to the

receptors. However, rather than selecting a single representative exposure scenario, the exposure models will be solved probabilistically over the entire distributions of observed CoPC concentrations and projected receptor population distributions to provide estimates of the probability of a given exposure rate over all possible outcomes. The general structure of the exposure algorithm used to estimate the exposure rate and the probability of exposure is as follows:

$$EED_{(x,y,t)} = IR_p \times \int \left([CoPC]_p\right)_{x,y,t} + IR_w \times \int \left([CoPC]_w\right)_{x,y,t} + IR_s \times \int \left([CoPC]_s\right)_{x,y,t}$$

$$p(EED) \equiv \frac{n(EED)_i}{\int_{i=1}^{\infty} (EED)_i} \quad \text{Where} : \sum p(EED) = 1$$

where:

IR_p = distribution of receptor-specific prey intake rate (kg dry weight/kg body weight-day)

IR_w = distribution of receptor-specific water intake rate (L/kg body weight-day)

IR_s = distribution of receptor-specific incidental soil intake rate (kg dry weight/kg body weight-day)

$[CoPC]_p$ = specific CoPC concentrations in the receptor's prey (mg/kg dry weight)

$[CoPC]_w$ = specific CoPC concentrations in the receptor's drinking water (mg/L)

$[CoPC]_s$ = specific CoPC concentrations in the sediments or soil incidentally ingested (mg/kg dry weight)

$EED_{(x,y,t)}$ = estimated environmental dose for a specific location and duration (mg/kg body weight-day)

EED_{Total} = estimated environmental dose derived for an individual of a receptor population (mg/kg body weight-day)

p(EED) = probability of exposure to a specific estimated environmental dose (unitless)

n(EED) = frequency of exposure to a specific estimated environmental dose (unitless).

Integration of exposure pathways for exposed receptor populations will be determined through summation of all possible exposure outcomes (e.g., at various places and times), based on the probability that a quantified proportion of the exposed populations would experience such conditions. This integration will be simulated using Monte Carlo analysis to provide a probabilistic evaluation of total exposure rate for the modeled receptor populations.

Site-specific data on the distribution of chemical concentrations in abiotic media and key prey species will be incorporated directly into the model along with assumptions regarding behavioral and life history aspects of the receptors (i.e., food ingestion rates, foraging range, habitat preference, and migration patterns) to estimate CoPC exposure distributions. Natural

history information is compiled from literature citations for each receptor. Exposure parameters and distributions for each species are provided in Tables 5-3 through 5-10. Depending on the results of deterministic food web models, some or all of these receptors may not require evaluation using probabilistic methods.

The distributions of CoPC exposures will be based on the concentration and frequency of observations in samples taken from the study area by developing distributions of CoPC concentrations as measured onsite. The distributions will be resolved over 10,000 iterations using Monte Carlo simulations, and frequency distributions will then be used as CoPC concentration inputs to the food web models.

Exposure to media associated with the study area is proportional to the projected residency period of the receptor in the study area. Migratory species that do not spend the entire year within the study area (e.g., caribou) will be assumed to be exposed to ambient levels of CoPCs measured in media and prey at the reference areas for that duration of the year when they are not present at the study area. Concurrent with the evaluation of risk to populations most exposed to the DMTS system, an evaluation of risk resulting from reference area exposures will also be determined to provide a context for the magnitude of the effect attributable to the study area. The risk attributable to the study area, and thereby that affected by any potential remedial activities specific to the DMTS, will be the difference between the proportion of the population affected by exposure to the DMTS area and the proportion of the population affected by exposure to reference area conditions.

5.5 Toxicity Assessment

To evaluate the potential for adverse effects to avian and mammalian ecological receptors, exposure estimates will be calculated and compared to LOAEL-based TRVs as described above. A TRV is a body-weight-normalized daily intake rate of a chemical that, if exceeded, could potentially result in adverse effects to the ecological receptor. The selection of TRVs requires the use of professional judgment in combination with guidelines provided in EPA's ERA guidance documents. Because the intent of an ERA is to assess risk to wildlife populations (U.S. EPA 1997a), exposure studies are evaluated for the endpoints that affect receptors on a population level: development, reproduction, and survival. Chronic dietary exposure studies are preferred, because they best represent wildlife exposure conditions to CoPCs, but for some chemicals, little or no toxicological information has been published. TRVs are expressed in body-weight-normalized units of mg/kg-day, which enables application of the TRV to various species consuming diets of variable moisture content. Table 3-28 presents the avian and mammalian TRVs that will be used in the baseline assessment. Brief technical reviews of the studies used in derivation of individual TRVs will be presented in the *Exposure Evaluation* section of the baseline risk assessment.

Allometric scaling of TRVs will not be performed in this ERA. Sample and Arenal (1999) examined allometric models for interspecies extrapolation of TRVs. Although they determined mean scaling values of 1.2 and 0.94 for birds and mammals, respectively, many chemical-specific scaling factors did not differ significantly from 1. For the limited set of metal TRVs presented in Sample and Arenal (1999), 21 of 24 scaling factors did not differ significantly

from 1. Furthermore, scaling factors presented in Sample and Arenal (1999) are all based on acute toxicity data, and as the authors note, their applicability to chronic toxicity data is unknown, and different scaling factors would need to be developed to allometrically scale chronic TRVs. Therefore, because there is no strong evidence for application of scaling factors other than 1 for chronic avian or mammalian metal TRVs, no scaling factors will be used in this ERA.

5.6 Risk Characterization

The estimation of risks to ecological receptors will be based on an integration of the exposure and effects assessments. For aquatic and terrestrial plants, aquatic and terrestrial invertebrates, and for fish, this integration will entail comparison of measured CoPC concentrations in environmental media and/or tissue to literature effects levels. For avian and mammalian receptors, the modeled distributions of exposure will be based on the observed distributions of CoPC concentrations in environmental media and prey and the quantitative characteristics of the target receptor populations. Exposure distributions will be used to develop a metric of the expected frequency of impacts related directly to exposures to CoPCs at the site. The risks estimated for various ecological receptors will be integrated and interpreted to evaluate their overall significance to the study area ecosystems, and to help identify what remedial actions, if any, may be required to reduce these risks.

5.7 Uncertainty Assessment

The risk characterization will include a detailed evaluation of sources of uncertainty and the effects of these uncertainties on conclusions about the extent and magnitude of risks. There are likely to be several major sources of uncertainty related to results of the DMTS ERA, which may include, but are not necessarily restricted to, these listed below:

- Evaluation of potential risks related to water
  - Representativeness of sampling locations
  - Comparisons with water quality values
  - Uncertainty in correlating observed aquatic community responses with CoPC concentrations in water
  - Uncertainty in extrapolation of risks to aquatic populations.
- Evaluation of potential risks related to sediment
  - Representativeness of sampling locations
  - Comparisons with sediment quality values

  - Uncertainty in correlating observed aquatic plant and benthic community responses with CoPC concentrations in sediment
  - Uncertainty in extrapolation of risks to aquatic plant and benthic invertebrate populations.

- Evaluation of potential risks related to soil
  - Representativeness of sampling locations
  - Comparisons with soil toxicity benchmarks
  - Uncertainty in correlating observed terrestrial plant and soil fauna community responses with CoPC concentrations in tundra soil.
  - Uncertainty in extrapolation of risks to terrestrial vegetation and soil fauna populations.
- Evaluation of potential risks to wildlife
  - Wildlife exposure estimates
  - TRVs
  - Uncertainty in TRV extrapolation
  - Population-level uncertainty
  - Uncertainty in risk characterization.

Major sources of uncertainty and their effects on risk characterization conclusions will be discussed in detail in the ERA.

6 Calculation of Risk-Based Action Levels

The risk assessment process defined in the DEC risk assessment procedures manual (DEC 2000) and 18 AAC 75.340 provides for the calculation of site-specific risk-based alternative cleanup levels (alternative to the default DEC cleanup levels) if site conditions are not "protective of human health, safety, and welfare, and of the environment," as indicated by a site-specific risk assessment. However, because the DMTS is an active facility (rather than a closed facility typically dealt with by the contaminated sites program guidance), it is more appropriate to refer to these values as "action levels" rather than "cleanup levels." In most cases, establishing action levels is more appropriate because, although active efforts are being made to control and minimize fugitive dust generation, some level of ongoing dust deposition is expected over the life of the mine. Action levels, if exceeded on a temporal or spatial basis, could trigger additional evaluation and implementation of risk management, control, or monitoring activities, as illustrated in Figure 1-1, the decision-making framework from DEC et al. (2002). In areas immediately adjacent to facilities where some higher concentrations are present (e.g., at the port), the action levels may function more as traditional "cleanup levels."

6.1 Human Health Based Action Levels

If the DMTS HHRA concludes that unacceptable risks exist at the site, risk-based action levels will be calculated for facility and road areas through application of the following general algorithm, which solves for a concentration through application of the exposure terms described in Section 4.2, the toxicity values described in Section 4.3, and a target risk level. The general algorithm for chemicals other than lead is as follows:

$$\text{Risk - Based Action Level (mg/kg)} = \frac{\text{body weight} \times \text{averaging time} \times \text{toxicity value}}{\text{target risk} \times \text{exposure term}}$$

where:

body weight	=	body weight of the receptor in kg
averaging time	=	averaging time in days
toxicity value	=	RfD
target risk	=	hazard index of 1
exposure term	=	exposure terms as described in Section 4.2.

The specific exposure terms, which include exposure duration, exposure frequency, and fractional intake, are described in Section 4.2 of this document.

In deriving a risk-based action level for lead, available default soil cleanup levels will first be considered. To develop site-specific action levels, the IEUBK and/or the adult lead

methodology will be run with successively higher assumed soil concentrations to determine an exposure level that meets applicable criteria, as described in Section 4.3.

6.2 Ecological Risk Based Action Levels

DEC guidance has no specific guidelines on how to develop ecological risk-based cleanup levels. In some cases, (e.g., for plant communities), the results of community surveys may be used to identify appropriate action levels. In other cases, the question of what is unacceptable risk in an ecological context may be established by defining a proportion of a receptor population for which an impact would result in population instability for a specific receptor or any subsequent trophic level dependent upon the specific receptor population assessed. Once such an endpoint has been established, then it will be possible to use the exposure distributions developed in the probabilistic risk assessment to derive strategies to ensure that the presence of CoPCs will not adversely affect the ecological structure of the assessed regions.

7 Conclusions and Data Needs

The next step in the risk assessment will be the preparation of a field sampling plan to address the additional data needs identified during the course of the work to date. The ERA will require biota sample collection to obtain data for food or prey items associated with the receptors. Table 7-1 summarizes the ERA data needs in relation to each environment, assessment endpoint, receptor, and associated food item. Additional data needs for the HHRA include the collection of ptarmigan (a subsistence food item) to be analyzed for barium, cadmium, lead, and zinc. Further details on the collection and analysis of samples to address the HHRA and ERA data needs will be provided in the field sampling plan, which will describe work to be conducted during the summer 2004 field season.

In the risk assessment document, the *Conclusions* section will review the risk assessment results in a spatial context, summarizing areas of the site that may pose an unacceptable risk. Following completion of the risk assessment, DEC will determine what actions may be needed to address any risks that are identified. This stage of the process includes evaluation and implementation of risk management, control, and monitoring options, as illustrated in Figure 1-1, the decision-making framework from DEC et al. (2002).

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Tables

Table 2-1. Composition of Red Dog lead and zinc concentrates

	Concentrate	
Element/Compound	Lead	Zinc
Aluminum oxide[a]	2,600	1,300
Antimony[a]	1,600	400
Arsenic[a]	400	200
Barium[a]	2,400	2,700
Cadmium[a]	1,200	3,300
Calcium oxide[a]	600	500
Copper[a]	600	1,400
Iron[a]	53,000	50,000
Lead[a]	579,000	32,000
Manganese oxide[a]	100	100
Silicon oxide[a]	38,000	35,000
Sulfate[a]	4,000	4,500
Sulfur (total)[a]	205,000	317,000
Zinc[a]	108,000	552,000
Bismuth	6	8
Chloride	50	50
Chromium	677	537
Cobalt	77	98
Fluoride	64	56
Gallium	11	26
Germanium	17	79
Gold	0.22	0.209
Manganese	12	10
Mercury	18	94
Molybdenum	43	20
Nickel	45	16
Selenium	28	3
Silver	420	137
Strontium	11	10
Thallium	70	19
Tin	23	49
Vanadium	23	13

Source: Teck Cominco (2002)

Note: All units are expressed in ppm (i.e., mg/kg dry weight basis).

[a] Values are based on dry weight; actual lead and zinc concentrates contain approximately 8.5 and 9.5 percent water, respectively.

Table 2-2. DMTS-related spills from DEC database

Spill Number	Spill Date	Spill Name	Quantity Released	Quantity Unit ID	Substance Type	Clean Up
93389921001	07/29/93	36,000 gal diesel (Tank #2)	36,000	gallons	Diesel	yes
98389915802	06/07/98		70	gallons	Diesel	yes
98389933101	11/27/98		55	gallons	Diesel	yes
97389925701	09/14/97		30	gallons	Diesel	yes
98389915702	06/06/98		20	gallons	Diesel	yes
98389908201	03/23/98		15	gallons	Diesel	yes
99389935401	12/20/99		10	gallons	Diesel	yes
00389907901	03/19/00		10	gallons	Diesel	yes
00389923201	08/19/00		10	gallons	Diesel	yes
96389929601	10/22/96		15	gallons	Engine lube oil	yes
98389921901	08/07/98		10	gallons	Engine lube oil	yes
01389919601	07/15/01		90	gallons	Hydraulic oil	yes
98389921601	08/04/98		45	gallons	Hydraulic oil	yes
01389903101	01/31/01		30	gallons	Hydraulic oil	yes
97389917301	06/22/97		25	gallons	Hydraulic oil	yes
00389915601	06/04/00		25	gallons	Hydraulic oil	yes
97389922401	08/12/97	Port old CSB-column 47	20	gallons	Hydraulic oil	yes
99389933702	12/03/99		20	gallons	Hydraulic oil	yes
95389934901	12/15/95		20	gallons	Hydraulic oil	no
97389931801	11/14/97		15	gallons	Hydraulic oil	yes
96389933401	11/29/96		10	gallons	Hydraulic oil	yes
98389915203	06/01/98		10	gallons	Hydraulic oil	yes
97389927803	10/05/97		65	gallons	Other[a]	yes
98389921301	08/01/98	Port CSB	76,000	pounds	Lead concentrate	yes
99389902101	01/21/99	Port Road mile 9	60,000	pounds	Lead concentrate	yes
00389928301	10/09/00	Port Road mile marker 31	52,000	pounds	Lead concentrate	yes
99389910101	04/11/99	Port CSB	150	pounds	Lead concentrate	yes
98389931402	11/10/98		200	pounds	Other[b]	yes
98389903801	02/07/98	Zinc at mile marker 27.25	140,000	pounds	Zinc concentrate	yes
00389936301	12/28/00	Port Road mile marker 45	88,100	pounds	Zinc concentrate	yes
97389923301	08/21/97	Port site entrance to racetrack	70,000	pounds	Zinc concentrate	yes
98389932501	11/21/98	Port Road mile 41.75 by MS 11	70,000	pounds	Zinc concentrate	yes
99389900601	01/06/99	Port Road mile 45	50,000	pounds	Zinc concentrate	yes
97389900201	01/02/97	40,000 lb zinc at mile marker 27	40,000	pounds	Zinc concentrate	yes
98389901701	01/17/98	Port Road mile 35 (near MS9)	37,760	pounds	Zinc concentrate	yes
98389919301	07/12/98	Mile post 42	26,500	pounds	Zinc concentrate	yes
96389915901	06/07/96	6,743 lb zinc conc. at MS-2	2,000	pounds	Zinc concentrate	yes
98389910701	04/17/98	Port Road 150 ft south of Tutak Bridge	800	pounds	Zinc concentrate	yes
00389923402	08/21/00	Port - Conveyor P-10 drive house	750	pounds	Zinc concentrate	yes
01389920101	07/20/01	Red Dog Mine zinc spill MP 38.3	20,000	pounds	Zinc concentrate	yes

Note: CSB - concentrate storage building
DEC - Alaska Department of Environmental Conservation

Data were provided by DEC from its Prevention and Early Response & Preparation database.
Table includes spills greater than or equal to 10 gallons or 10 pounds.
Database does not include spills that occurred prior to 1995, except for July 29, 1993, spill.

[a] Uncertain, but possible match with process water spill at the mine mill on October 5, 1997.

[b] Uncertain, but possible match with 1 gallon spill of ethylene glycol at the mine (Hagy 2003, pers. comm.)

Table 2-3. Relative importance of potential human exposure pathways[a]

Metal	Human Exposure Pathways			Cumulative PRG
	Inhalation	Dermal	Ingestion	
Aluminum	2,882,040		78,214	76,142
Antimony			31	31
Arsenic	588	4	0.4	0.4
Barium	294,086		5,475	5,375
Cadmium (cancer)	1,405			1,404
Cadmium (noncancer)		698	39	37
Chromium VI (cancer)	30			30
Chromium VI (noncancer)	4,529		235	223
Cobalt (cancer)	903			903
Cobalt (noncancer)	11,734		1,564	1,380
Copper			3,129	3,129
Fluoride		16,760	4,693	3,666
Iron			23,464	23,463
Lead				
Manganese	28,820		1,877	1,762
Mercury			23	23
Molybdenum			391	391
Nickel			1,564	1,564
Selenium			391	391
Silver			391	391
Strontium			46,929	46,924
Thallium			5	5
Tin			46,929	46,924
Vanadium			548	547
Zinc			23,464	23,463

Note: Units are in mg/kg.
EPA - U.S. Environmental Protection Agency
PRG - preliminary remediation goal

[a] The screening values listed above are U.S. EPA (2003c) Region 9 PRGs for residential soil. This table is not meant to provide screening concentrations for the DMTS risk assessment. Rather, the PRGs listed above are provided to illustrate the relative contribution of inhalation, dermal contact, and ingestion exposure. The PRGs were derived assuming a risk level of $1x10^{-6}$ for cancer and a hazard quotient of 1.0 for noncancer endpoints. Higher PRGs indicate relatively lower contribution to risk, and vice versa. These PRGs suggest that dermal contact is at least an order of magnitude lower risk than ingestion, and that inhalation is several orders of magnitude lower risk than ingestion.

Table 2-4. Subsistence resource categories and representative receptors

Subsistence Resource[a]	Inupiat Name[b]	Scientific Name	Representative Receptors
Fish			**Fish**
Bluecod			**Note:** Risk to fish evaluated by comparison of chemical concentrations in sediment and water to effects ranges reported in the literature
Bullhead	kanayuq	*Myoxocephalus quadricornis*	
Burbot/mudshark	tittaaliq	*Lota lota*	
Char (arctic)	aqalukpik	*Salvelinus alpinus*	
Dolly Varden		*Salvelinus malma*	
Grayling (arctic)	suluppaugaq	*Thymallus arcticus*	
Herring (Pacific)	uqsruqtuuq	*Clupea pallasi*	
Pike (northern)	siilik	*Esox lucius*	
Salmon (king)		*Oncorhynchus tshawytscha*	
Salmon (pink)	qalugruaq	*Onchorynchus gorbuscha*	
Salmon (silver)		*Oncorhynchus kisutch*	
Sheefish	sii	*Stenodus leucichthys*	
Smelt (rainbow)	ijhuabniq	*Osmerus mordax dentex*	
Tomcod (arctic cod)	uugaq	*Boreogadus saida*	
Whitefish	qalupiat, quptik, qaalbiq, qalusraaq	*Coregonis* spp., *Prosopium* spp.	
Shellfish			**Shellfish**
Crab	putyuun	*Paralithodes* spp., *Lithodes aequispinus, Cancer magister, Chionoecetes* spp.	**Note:** Risk to shellfish evaluated by comparison of chemical concentrations in sediment and water to effects ranges reported in the literature
Shrimp	putuguqsiuyuk	*Pandalus* spp., *Pandalopsis* spp.	
Sea Mammals			**Sea Mammals**
Seal (bearded)	ugruk	*Erignathus barbatus*	Bearded seal (invertebrate-eater)
Seal (ribbon)	qaigullik	*Phoca fasciata*	Ringed seal (fish- and invertebrate-eater)
Seal (spotted)	qasigiaq	*Phoca largha*	
Walrus	aiviq	*Odobenus rosemarus*	
Whale (Beluga)	sisuaq	*Delphinapterus leucas*	
Whale (bowhead)	abviq	*Balaena mysticetus*	
Large Mammals			**Large Mammals**
Bear (black)	iyyabriq	*Ursus americanus*	Caribou (plant-eater)
Bear (brown/grizzly)	akjaq	*Ursus horribilis*	Moose (plant-eater)
Caribou	tuttu	*Rangifer tarandus*	Polar bear (marine animal-eater)
Moose	tiniikaq	*Alces alces*	
Muskox	imummak	*Ovibus moschatus*	
Sheep (dall)	ipnaiq	*Ovis dalli dalli*	
Small Mammals			**Small Mammals**
Beaver	aqu	*Castor canadensis*	Arctic fox (terrestrial animal-eater)
Fox (arctic)	qujhaaq	*Alopex lagopus*	Muskrat (freshwater plant-eater)
Fox (red)	kavviaq	*Vulpes fulva*	River otter (freshwater fish-eater)
Muskrat	kigvaluk	*Ondatra zibethicus*	Tundra shrew (terrestrial invertebrate-eater)
Otter (river)	pamiuqtuuq	*Lutra canadensis*	Tundra vole (terrestrial plant-eater)
Porcupine	ixuqutaq	*Erethizon dorsatum*	
Rabbit (Alaska hare)	ukallisugruk	*Lepus othus*	
Rabbit (snowshoe hare)	ukalliuraq	*Lepus americanus*	
Squirrel (ground)	siksrik	*Citellus parryi*	
Squirrel (red/tree)	saqalataayiq	*Tamiasciurus hudsonicus*	
Wolverine	qapvik	*Gulo luscus*	
Birds			**Birds**
Crane (sandhill)	tatirgaq	*Grus canadensis*	Black-bellied plover (marine insect-eater)
Ducks	qaugak	Multiple species	Brant (marine plant-eater)
Geese		Multiple species	Common snipe (freshwater invertebrate-eater)
Grouse (spruce)		*Falcipennis canadensis*	Green-winged teal (freshwater plant-eater)
Owl (snowy)	ukpik	*Nyctea scandiaca*	Lapland longspur (terrestrial insect-eater)
Ptarmigan (mountain)			Red-throated loon (freshwater and marine fish-eater)
Ptarmigan (willow)	aqargiq	*Lagopus lagopus*	Snowy owl (terrestrial animal-eater)
Swans (tundra, formerly whistling)	qugruk	*Cygnus columbianus*	Willow ptarmigan (terrestrial plant-eater)

Table 2-4. (cont.)

Subsistence Resource[a]	Inupiat Name[b]	Scientific Name	Representative Receptors
Vegetation			**Vegetation**
Blueberry	asriavik	*Vaccinium uliginosum*	**Note:** Risk to plants evaluated by comparison of chemical concentrations in soil and plants to effects ranges reported in the literature
Coast greens sura (fresh willow leaf)		*Salix* spp.	
Cow parsnip		*Heracleum lanatum (Umbelliferae)*	
Cranberry (bog)	qunmun asriaq	*Vaccinium oxycoccus*	
Cranberry (highbush)	uqpifeaq	*Viburnum edule*	
Cranberry (lowbush)	kikmieeaq	*Vaccinium vitis idaea*	
Crowberry (black berry)	paunbaq	*Empetrum nigrum*	
Eskimo (labrador/tundra) tea		*Ledum decubens*	
Eskimo/wild potato	masru	*Hedysarum alpinum*	
Herbal tea		Species unknown	
Matsu sura		Species unknown	
Raspberry	tuunbaum asriaq	*Rubus pendantus*	
Salmonberry/cloudberry	aqpik	*Rubus chamaemorus*	
Sourdock	quabaq	*Rumex arcticus*	
Spring tea		Species unknown	
Stinkweed		*Thlaspi arvense*	
Wild celery	ikuusuk	*Angelica lucida*	
Wild onions/chives	paatitaaq	*Allium schoenoprasum sibiricum*	
Wild rhubarb	qusrimmak	*Polygonum alaskanum*	
Wild tea		Species unknown	

[a] Sundet (2002a,b, pers. comm.).

[b] Webster and Zibell (2003).

Table 2-5. Summary of preliminary assessment endpoints, representative receptors, and measurement endpoints[a]

Environment	Assessment Endpoint	Representative Receptor	Measurement Endpoint
Terrestrial	Structure and function of **terrestrial plant communities**	Terrestrial plant communities	Range of CoPC concentrations in soil relative to ecological screening benchmarks
Terrestrial	Structure and function of **terrestrial fauna communities**	Soil fauna communities	Range of CoPC concentrations in soil relative to ecological screening benchmarks
Terrestrial	Survival, growth, and reproduction of **terrestrial avian herbivore populations**	Willow ptarmigan	Range of modeled total dietary exposures relative to avian TRVs
Terrestrial	Survival, growth, and reproduction of **terrestrial mammalian herbivore populations**	Tundra vole; caribou; moose	Range of modeled total dietary exposures relative to mammalian TRVs
Terrestrial	Survival, growth, and reproduction of **terrestrial avian invertevore populations**	Lapland longspur	Range of modeled total dietary exposures relative to avian TRVs
Terrestrial	Survival, growth, and reproduction of **terrestrial mammalian invertevore populations**	Tundra shrew	Range of modeled total dietary exposures relative to mammalian TRVs
Terrestrial	Survival, growth, and reproduction of **terrestrial avian carnivore populations**	Snowy owl	Range of modeled total dietary exposures relative to avian TRVs
Terrestrial	Survival, growth, and reproduction of **terrestrial mammalian carnivore populations**	Arctic fox	Range of modeled total dietary exposures relative to mammalian TRVs
Freshwater Aquatic	Structure and function of **freshwater aquatic and wetland plant communities**	Freshwater aquatic and wetland plant communities	Range of CoPC concentrations in freshwater sediment and water relative to ecological screening benchmarks
Freshwater Aquatic	Structure and function of **freshwater aquatic invertebrate communities**	Freshwater aquatic invertebrate communities	Range of CoPC concentrations in freshwater sediment relative to ecological screening benchmarks
Freshwater Aquatic	Structure and function of **freshwater fish communities**	Freshwater fish communities	Range of CoPC concentrations in freshwater relative to ecological screening benchmarks
Freshwater Aquatic	Survival, growth, and reproduction of **freshwater avian herbivore populations**	Green-winged teal	Range of modeled total dietary exposures relative to avian TRVs
Freshwater Aquatic	Survival, growth, and reproduction of **freshwater mammalian herbivore populations**	Muskrat	Range of modeled total dietary exposures relative to mammalian TRVs
Freshwater Aquatic	Survival, growth, and reproduction of **freshwater avian invertevore populations**	Common snipe	Range of modeled total dietary exposures relative to avian TRVs

Table 2-5. (cont.)

Environment	Assessment Endpoint	Representative Receptor	Measurement Endpoint
Freshwater Aquatic	Survival, growth, and reproduction of **freshwater avian piscivore populations**	Red-throated loon	Range of modeled total dietary exposures relative to avian TRVs
Freshwater Aquatic	Survival, growth, and reproduction of **freshwater mammalian piscivore populations**	River otter	Range of modeled total dietary exposures relative to mammalian TRVs
Marine	Structure and function of **marine aquatic and wetland communities**	Marine aquatic and wetland plant communities	Range of CoPC concentrations in marine sediment and water relative to ecological screening benchmarks
Marine	Structure and function of **marine aquatic invertebrate communities**	Marine aquatic invertebrate communities	Range of CoPC concentrations in marine sediment and water relative to ecological screening benchmarks
Marine	Structure and function of **marine fish communities**	Marine fish communities	Range of CoPC concentrations in marine water relative to ecological screening benchmarks
Marine	Survival, growth, and reproduction of **marine avian herbivore populations**	Brant	Range of modeled total dietary exposures relative to avian TRVs
Marine	Survival, growth, and reproduction of **marine avian invertevore populations**	Black-bellied plover	Range of modeled total dietary exposures relative to avian TRVs
Marine	Survival, growth, and reproduction of **marine mammalian invertevore populations**	Bearded seal	Range of modeled total dietary exposures relative to mammalian TRVs
Marine	Survival, growth, and reproduction of **marine avian piscivore populations**	Red-throated loon	Range of modeled total dietary exposures relative to avian TRVs
Marine	Survival, growth, and reproduction of **marine mammalian piscivore populations**	Ringed seal	Range of modeled total dietary exposures relative to mammalian TRVs
Marine	Survival, growth, and reproduction of **marine mammalian carnivore populations**	Polar bear	Range of modeled total dietary exposures relative to mammalian TRVs

Note: CoPC - chemical of potential concern
TRV - toxicity reference value

[a] A refined version of this table was developed following CoPC screening and is presented later in this document.

Table 3-1. Target chemical list

Aluminum
Antimony
Arsenic
Barium
Cadmium
Chromium
Cobalt
Copper
Fluoride
Iron
Lead
Manganese
Mercury
Molybdenum
Nickel
Selenium
Silver
Strontium
Thallium
Tin
Vanadium
Zinc

Table 3-2. Overview of prior studies

Lead Organization	Study Type	Citation	Study Dates	Analytical Data Available						
				Moss	Soil	Water	Sediment	Plants	Fish	Caribou
Pre-Mine/Baseline										
Teck Cominco	Environmental baseline study	Dames & Moore (1983a,b)	1981–1983			●	●		●	
General Crude Oil and Minerals	Environmental baseline study	Ward and Olson (1980)	1978–1979			●			●	
Alaska Department of Environmental Conservation	Aquatic baseline study	EVS and Ott Water (1983)	1982						●	
U.S. Fish and Wildlife Service	Baseline study for Selawik NWR	Mueller et al. (1993)	1987–1988			●	●		●	
Post-Mine										
Teck Cominco	Port site monitoring	ENSR (1990, 1991, 1993, 1996); RWJ (1997)	1990–1996		●	●	●			
	Transportation corridor monitoring	ENSR (1991)	1991–1992		●	●				
	Vegetation and soil monitoring	RWJ (1998)	1992, 1993, 1997		●					
	Fugitive dust study	Exponent (2002a); DEC et al. (2002)	2001	●	●	●		●		
	Kivalina drinking water study	RWJ (1997); DHSS (2001); (Kulas 2003, pers. comm.)	1991–2003			●				
	Supplemental road sampling	Exponent (2002b)	2002		●					
	Caribou evaluation	Exponent (2002c)	1996, 2002							●
	Port site characterization	Exponent (2003a)	2002		●	●	●			
	Phase I risk assessment field sampling program	Exponent (2003f) and Appendix A of this document	2003	●	●	●	●			
Alaska Industrial Development and Export Authority	Sediment quality survey	Cominco et al. (1999)	1998		●		●			
Alaska Department of Environmental Conservation	Subsistence foods investigation	E&E (2002); DHSS (2001)	2001			●		●		
Alaska Department of Fish and Game	NPDES monitoring, expanded scope	Weber-Scannell and Ott (2001)	1994–2001			●			●	
	Juvenile fish tissue study	Morris and Ott (2001); DHSS (2001)	1993, 1998–2001						●	
National Park Service	DMTS road dustfall study	Ford and Hasselbach (2001)	2000	●	●					
		Hasselbach (2003a, pers. comm.)	2003	●	●					
Kivalina Village	Kivalina drinking water sampling	DHSS (2001)	1995, 1996, 2001			●				
United States Geological Survey	Cape Krusenstern trace elements study	Brabets (2003, pers. comm.)	2002			●	●			
	Willow study	Gough (2003, pers. comm.)	2002					●		
	Soil study	Kelley and Hudson (2003)	2002		●					

Table 3-3. Analytical data summary for screening chemicals of potential concern

Environment	Medium	Site/ Reference	Survey Name	Numbers of Samples by Analyte[a]																					
				Aluminum	Antimony	Arsenic	Barium	Cadmium	Chromium	Cobalt	Copper	Fluoride	Iron	Lead	Manganese	Mercury	Molybdenum	Nickel	Selenium	Silver	Strontium	Thallium	Tin	Vanadium	Zinc
Terrestrial	Soil	Site	PHASE1RA, PSCHAR, FUGDST01, SUPPRSS, TECK03	51	40	75	40	478	40	40	40	12	51	479	40	12	40	40	30	40	20	12	27	40	479
		Reference	PHASE1RA, FUGDST01	10	5	10	5	10	5	5	5	5	10	10	5	5	5	5	5	5	5	5	5	5	10
	Tundra Soil	Site	PHASE1RA, PSCHAR, ENSR92	31	25	31	25	224	25	25	25	12	31	264	25	12	25	25	25	25	17	12	17	25	264
		Reference	PHASE1RA	10	10	10	10	10	10	10	10	10	10	10	10	10	10	10	10	10	10	10	10	10	10
Stream	Sediment	Site	PHASE1RA	14	14	14	14	14	14	14	14	14	14	14	14	14	14	14	14	14	14	14	14	14	14
		Reference	PHASE1RA	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5
	Water	Site	TECK03, TECK01, USGS02	230	14	14	14	229	18	14	18	31	230	230	18	14	14	14	29	14	14	29	14	14	230
		Reference	PHASE1RA	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3
Tundra Pond	Sediment	Site	PHASE1RA	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4
		Reference	PHASE1RA	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5
	Water	Site	PHASE1RA	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4
		Reference	PHASE1RA	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3
Lagoon	Sediment	Site	PHASE1RA, PSCHAR	8	8	8	8	34	8	8	8	11	8	26	8	8	8	8	8	8	8	8	11	8	26
		Reference	PHASE1RA, PSCHAR, ENSR91, ENSR92, ENSR95, ENSR96	3	3	3	3	13	3	3	3	3	3	28	3	3	3	3	3	3	3	3	3	3	28
	Water	Site	PHASE1RA, PSCHAR	8	8	8	8	14	8	8	8	8	8	14	8	8	8	8	8	8	8	8	8	8	14
		Reference	PHASE1RA, PSCHAR	3	3	3	3	5	3	3	3	3	3	5	3	3	3	3	3	3	3	3	3	3	5
Marine	Sediment	Site	PHASE1RA, PSCHAR, CORPS00, DMTP98	18	17	69	69	129	69	18	69	16	18	129	18	16	18	18	17	69	17	17	17	41	129
		Reference	PHASE1RA, DMTP98, BASLIN82	15	9	21	21	21	21	9	21	9	15	21	9	9	9	15	15	21	9	9	9	9	21
	Water	Site	PHASE1RA	9	9	9	9	9	9	9	9	9	9	9	9	9	9	9	9	9	9	9	9	9	9
		Reference	PHASE1RA	6	6	6	6	6	6	6	6	6	6	6	6	6	6	6	6	6	6	6	6	6	6

Note: Survey names and citations:

PHASE1RA	Exponent (2003f) and Appendix A of this document
PSCHAR	Exponent (2003c)
FUGDST01	Exponent (2002a)
SUPPRSS	Exponent (2002b)
TECK03	Teck Cominco (2003)
TECK01	Exponent (2002a)
ENSR91	ENSR (1992)
ENSR92	ENSR (1993)
ENSR95	ENSR (1996)
ENSR96	RWJ (1997)
DMTP98	Cominco et al. (1999)
CORPS00	Corps (2001)
BASLIN82	Dames & Moore (1983a)
USGS02	Brabets (2003, pers. comm.)

[a] The numbers of samples shown are for the data to be used in the assessment, processed according to the data usability criteria listed in Section 3.2 of the main text.

Table 3-4. Statistical comparison of site and reference soil data

	Reference					Site					Site >
Chemical	N	Min	Max	Mean	Stdev	N	Min	Max	Mean	Stdev	Reference?[a]
Aluminum	10	1,640	12,400	6,963	4,351	51	1,180	16,600	7,392	3,281	no
Antimony											>50% ND
Arsenic	10	4.15	35	12.6	9.8	75	1.3	93.6	12.2	15.1	no
Barium	5	109	622	249	213	40	357	7,090	2,137	1,830	**yes**
Cadmium	10	0.24	3.59	1.1	1	478	0.4	388	25.2	37.8	**yes**
Chromium	5	4.94	19.3	11.8	5.7	40	4.86	24	15.0	5.1	no
Cobalt	5	7.28	20.6	13.5	5.1	40	4.21	27	11.3	5.0	no
Copper	5	14.3	46.5	23.7	13.0	40	9.76	109	36.0	20.3	no
Fluoride	5	0.3	0.5	0.42	0.08	12	0.4	1.3	0.73	0.30	**yes**
Iron	10	5,750	72,600	29,872	18,432	51	2,650	35,000	20,682	7,572	no
Lead	10	8.75	142	38.5	38.5	479	8.5	48,300	1,157	2,795	**yes**
Manganese	5	250	4,080	1,489	1,589	40	280	1,000	513	186	R>S
Mercury	5	0.05	0.18	0.11	0.053	12	0.1	1.69	0.45	0.51	**yes**
Molybdenum											>50% ND
Nickel	5	23.5	51.4	34.2	12.6	40	17.3	56.8	29.1	10.0	no
Selenium											>50% ND
Silver	5	0.05	0.25	0.13	0.089	40	0.14	8.3	2.2	2.0	**yes**
Strontium	5	9.3	63.6	31.0	21.2	20	36.2	90.1	63.2	15.5	**yes**
Thallium	5	0.101	0.236	0.16	0.055	12	0.112	1.32	0.47	0.36	**yes**
Tin											>50% ND
Vanadium	5	5.62	19.2	11.9	5.4	40	7.94	31.8	14.7	4.8	no
Zinc	10	72.5	753	181	204	479	37.4	64,300	4,140	6,201	**yes**

Note: CoPC - chemical of potential concern
ND - not detected

Concentrations are given in mg/kg dry weight.

Undetected values are treated as detections at one-half the detection limit for the statistical analysis. In cases where greater than or equal to 50 percent of the site values were undetected, statistical analyses were not performed. Further summary information is provided in the CoPC screening tables, including detection limits and detection frequencies.

Field replicates were averaged prior to statistical analysis.

[a] Results of statistical comparison. Bold indicates chemicals for which statistical testing showed site concentrations to be greater than reference concentrations at a significance level of alpha = 0.05.

Table 3-5. Statistical comparison of site and reference tundra soil data

	Reference					Site					Site >
Chemical	N	Min	Max	Mean	Stdev	N	Min	Max	Mean	Stdev	Reference?[a]
Aluminum	10	368	11,300	3,651	3,347	31	358	18,900	5,329	4,822	no
Antimony	10	0.11	0.28	0.17	0.062	25	0.15	25.8	6.2	6.1	no[b]
Arsenic	10	0.4	6.8	2.3	1.9	31	0.3	150	17.7	26.6	**yes**
Barium	10	108	624	315	196	25	53	5060	945	1,306	no[b]
Cadmium	10	0.12	0.88	0.3	0.22	224	0.3	258	15.3	31.7	**yes**
Chromium	10	1.57	19.7	6.8	6.1	25	1.03	33.2	10.4	8.7	no
Cobalt	10	0.96	28.3	8.6	10.6	25	0.5	35	11.0	9.3	no[b]
Copper	10	4.34	16.9	8.2	4.0	25	2.88	58.3	21.0	15.5	**yes**
Fluoride											>50% ND
Iron	10	912	45,100	12,909	13,600	31	593	181,000	26,417	35,855	no
Lead	10	2.9	23.3	8.9	6.7	264	7	16,000	665	1,816	**yes**
Manganese	10	33.5	6,620	918	2,013	25	28.6	3,400	825	882	no
Mercury	10	0.07	0.15	0.11	0.026	12	0.1	4.16	0.71	1.2	no[b]
Molybdenum	10	0.34	2.27	0.85	0.60	25	0.59	3.9	1.5	0.91	**yes**
Nickel	10	4.33	36.8	16.2	10.6	25	1.58	37.5	18.7	10.7	no
Selenium	10	0.4	1	0.56	0.22	25	0.3	29	8.9	9.0	**yes**
Silver	10	0.02	0.35	0.14	0.13	25	0.04	14.7	2.5	3.3	no[b]
Strontium	10	7.3	39.6	16.1	11.3	17	4.8	150	52.2	40.2	**yes**
Thallium	10	0.024	0.116	0.062	0.032	12	0.014	1.58	0.45	0.50	no[b]
Tin	10	2.1	17.4	5.350	4.600	17	2.1	15	7.55	4.48	no
Vanadium	10	1.3	24.7	9.7	7.6	25	0.7	46.5	14.5	12.4	no[b]
Zinc	10	47.8	111	66.1	24.2	264	22.3	48,700	2,127	4,880	**yes**

Note: CoPC - chemical of potential concern
ND - not detected

Concentrations are given in mg/kg dry weight.

Undetected values are treated as detections at one-half the detection limit for the statistical analysis. In cases where greater than greater than or equal to 50 percent of the site values were undetected, statistical analyses were not performed. Further summary information is provided in the CoPC screening tables, including detection limits and detection frequencies.

Field replicates were averaged prior to statistical analysis.

[a] Results of statistical comparison. Bold indicates chemicals for which statistical testing showed site concentrations to be greater than reference concentrations at a significance level of alpha = 0.05.

[b] Confidence interval for the site mean straddles zero, due to small sample size and/or high variability.

Table 3-6. Statistical comparison of site and reference stream sediment data

	Reference					Site					Site >
Chemical	N	Min	Max	Mean	Stdev	N	Min	Max	Mean	Stdev	Reference?[a]
Aluminum	5	3,620	12,100	6,848	3,652	14	4,080	17,100	7,846	3,560	no
Antimony	5	0.03	0.05	0.04	0.01	14	0.05	0.64	0.20	0.16	**yes**
Arsenic	5	3.5	8.1	5.1	1.8	14	3.3	11.4	7.8	2.1	**yes**
Barium	5	135	483	291	146	14	91.2	922	302	260	no
Cadmium	5	0.07	0.3	0.2	0.1	14	0.18	1.38	0.49	0.34	**yes**
Chromium	5	7.22	19.9	13	5	14	7.35	22.6	14.6	4.9	no
Cobalt	5	7.3	11	9.3	1.4	14	7.9	17.6	12.3	2.9	**yes**
Copper	5	5.99	18.5	11.3	4.6	14	9.66	28.2	15.9	4.8	**yes**
Fluoride											> 50% ND
Iron	5	21,300	27,300	24,500	2,279	14	22,800	45,700	30,479	5,898	**yes**
Lead	5	5.05	9.17	7.6	2	14	8.24	142	31.7	44.4	**yes**
Manganese	5	268	859	548	259	14	471	2140	995	542	**yes**
Mercury	5	0.005	0.04	0.027	0.015	14	0.02	0.089	0.044	0.020	no
Molybdenum	5	0.28	0.52	0.37	0.094	14	0.34	2.32	0.82	0.54	**yes**
Nickel	5	20.8	35	29.7	5.3	14	24.8	57.3	40.4	8.8	**yes**
Selenium	5	0.1	0.7	0.4	0.2	14	0.4	2.5	1.2	0.65	**yes**
Silver	5	0.03	0.12	0.1	0.0	14	0.05	0.42	0.19	0.12	**yes**
Strontium	5	4.9	15	12	4.1	14	11	155	45.5	42.6	**yes**
Thallium	5	0.023	0.07	0.050	0.019	14	0.031	0.322	0.10	0.076	**yes**
Tin	5	2	2.4	2.2	0.18	14	0.5	7.6	3.0	2.0	no
Vanadium	5	10.7	24.8	18	5.1	14	8.83	27.1	16.7	5.9	no
Zinc	5	43.7	69.7	62	11	14	58.4	259	139	52.0	**yes**

Note: CoPC - chemical of potential concern
ND - not detected

Concentrations are given in mg/kg dry weight.

Undetected values are treated as detections at one-half the detection limit for the statistical analysis. In cases where greater than or equal to 50 percent of the site values were undetected, statistical analyses were not performed. Further summary information is provided in the CoPC screening tables, including detection limits and detection frequencies.

Field replicates were averaged prior to statistical analysis.

[a] Results of statistical comparison. Bold indicates chemicals for which statistical testing showed site concentrations to be greater than reference concentrations at a significance level of alpha = 0.05.

Table 3-7. Statistical comparison of site and reference stream surface water data

	Reference					Site					Site >
Chemical	N	Min	Max	Mean	Stdev	N	Min	Max	Mean	Stdev	Reference?[a]
Aluminum	3	17.3	2,770	937	1,588	230	2.5	4,060	90.4	413	no
Antimony											>50% ND at site
Arsenic											100% ND at site
Barium	3	86.1	222	159	68.5	14	12.2	266	83.6	73.9	no
Cadmium											>50% ND at site
Chromium											100% ND at site
Cobalt	3	0.120	2.72	1.02	1.47	14	0.010	0.330	0.126	0.094	no
Copper	3	0.600	5.40	2.23	2.74	18	0.105	1.23	0.744	0.332	no
Fluoride	3	30.0	40.0	36.7	5.77	31	40.0	120	57.4	17.1	**yes**
Iron	3	64.2	6,710	2,295	3,823	230	2.57	10,300	318	1,078	no
Lead											>50% ND at site
Manganese	3	4.87	128	46.0	71.0	18	0.475	36.0	5.70	8.18	no
Mercury											100% ND at site
Molybdenum	3	0.02	0.170	0.080	0.079	14	0.178	2.27	0.739	0.685	**yes**
Nickel	3	1.06	10.5	4.46	5.25	14	0.260	6.71	2.07	1.80	no
Selenium	3	0.200	0.200	0.200	0	29	0.020	1.24	0.243	0	no; 100% ND for reference
Silver											100% ND at site
Strontium	3	32.5	81.1	54.9	24.5	14	19.4	172	83.0	55.5	no
Thallium											>50% ND at site
Tin											100% ND for reference, >50% ND at site
Vanadium											>50% ND at site
Zinc											>50% ND at site

Note: CoPC - chemical of potential concern
ND - not detected

Concentrations are given in μg/L unfiltered.

Undetected values are treated as detections at one-half the detection limit for the statistical analysis. In cases where greater than or equal to 50 percent of the site values were undetected, statistical analyses were not performed. Further summary information is provided in the CoPC screening tables, including detection limits and detection frequencies.

Field replicates were averaged prior to statistical analysis.

[a] Results of statistical comparison. Bold indicates chemicals for which statistical testing showed site concentrations to be greater than reference concentrations at a significance level of alpha = 0.05.

Table 3-8. Statistical comparison of site and reference pond sediment data

	Reference					Site					Site >
Chemical	N	Min	Max	Mean	Stdev	N	Min	Max	Mean	Stdev	Reference?[a]
Aluminum	5	3,730	17,100	9,908	5,750	4	1,920	4,330	3,288	1,214	R>S
Antimony	5	0.030	0.1	0.06	0.04	4	0.19	9.0	2.4	4.4	**yes**
Arsenic	5	2.6	13.0	6.6	4.2	4	2.6	7.5	4.7	2.1	no
Barium	5	121	772	430	257	4	281	498	372	95.3	R>S
Cadmium	5	0.27	0.66	0.4	0.16	4	0.93	101	26.2	49.9	**yes**
Chromium	5	9.6	28	19	8.6	4	9.0	13.0	10.3	1.8	no
Cobalt	5	1.8	21.9	10.3	7.4	4	2.7	24.1	14.2	10.7	no[b]
Copper	5	8.0	20.7	14.8	4.8	4	6.5	45.5	23.4	16.6	no
Fluoride											50% ND
Iron	5	17,900	43,700	27,140	11,232	4	16,000	51,900	29,300	15,682	no
Lead	5	7.48	20.3	11.6	5.0	4	9.0	1,810	484	885	no[b]
Manganese	5	15.9	1,870	515	766	4	79.8	745	290	312	no
Mercury	5	0.030	0.070	0.054	0.018	4	0.060	1.1	0.35	0.50	no[b]
Molybdenum	5	0.38	1.4	0.70	0.40	4	1.1	2.4	1.7	0.66	**yes**
Nickel	5	12.0	70.3	39.0	21.0	4	17.6	38.9	27.6	8.7	no
Selenium	5	0.50	3.1	1.2	1.09	4	0.75	3.0	1.6	0.97	no
Silver											50% ND
Strontium	5	4.2	25.4	12.2	8.1	4	17.1	86.0	37.5	32.5	no
Thallium	5	0.056	0.17	0.12	0.051	4	0.021	1.6	0.43	0.81	no
Tin											50% ND
Vanadium	5	14.9	94.5	41	31.129	4	12.2	28.3	17.8	7.2	no
Zinc	5	23.4	138	77	41.661	4	143	21,900	5,623	10,851	**yes**

Note: CoPC - chemical of potential concern
ND - not detected

Concentrations are given in mg/kg dry weight.

Undetected values are treated as detections at one-half the detection limit for the statistical analysis. In cases where greater than or equal to 50 percent of the site values were undetected, statistical analyses were not performed. Further summary information is provided in the CoPC screening tables, including detection limits and detection frequencies.

Field replicates were averaged prior to statistical analysis.

[a] Results of statistical comparison. Bold indicates chemicals for which statistical testing showed site concentrations to be greater than reference concentrations at a significance level of alpha = 0.05.

[b] Confidence interval for the site mean straddles zero, due to small sample size and/or high variability.

Table 3-9. Statistical comparison of site and reference pond surface water data

	Reference					Site					Site >
Chemical	N	Min	Max	Mean	Stdev	N	Min	Max	Mean	Stdev	Reference?[a]
Aluminum	3	14.5	170	91.9	77.8	4	11.4	177	102	73.6	no[b]
Antimony	3	0.020	0.10	0.057	0.040	4	0.020	0.20	0.085	0.083	no[b]
Arsenic	3	0.50	0.90	0.63	0.23	4	0.40	1.3	0.70	0.41	no
Barium	3	48.4	133	91.6	42.3	4	39.4	73.6	57.5	17.0	no
Cadmium	3	0.0050	0.060	0.038	0.0293	4	0.020	0.27	0.10	0.11	no[b]
Chromium	3	0.18	2.0	0.96	0.92	4	0.44	5.2	2.2	2.1	no[b]
Cobalt	3	0.19	0.70	0.37	0.29	4	0.13	1.6	0.76	0.61	no[b]
Copper	3	0.70	2.5	1.9	1.0	4	0.40	2.7	1.3	0.98	no
Fluoride	3	10.0	50.0	26.7	20.8	4	20.0	60.0	32.5	18.9	no
Iron	3	361	1,500	808	608	4	685	1,220	1,021	238	no
Lead	3	0.060	0.56	0.37	0.27	4	0.44	1.6	0.95	0.52	no
Manganese	3	4.2	71.2	32.1	34.9	4	2.9	132	53.5	58.5	no[b]
Mercury											>50% ND
Molybdenum	3	0.020	0.22	0.097	0.11	4	0.020	0.090	0.060	0.0316	no
Nickel	3	2.1	6.4	3.6	2.4	4	3.0	5.3	4.3	1.1	no
Selenium											>50% ND
Silver											>50% ND
Strontium	3	10.6	27.5	18.7	8.5	4	10.4	422	114	205	no
Thallium											>50% ND
Tin											>50% ND
Vanadium	3	0.17	2.4	1.2	1.1	4	0.24	0.64	0.37	0.18	no
Zinc	3	0.59	5.0	2.8	2.2	4	6.1	99.0	36.7	42.9	no[b]

Note: CoPC - chemical of potential concern
ND - not detected

Concentrations are given in μg/L unfiltered.

Undetected values are treated as detections at one-half the detection limit for the statistical analysis. In cases where greater than or equal to 50 percent of the site values were undetected, statistical analyses were not performed. Further summary information is provided in the CoPC screening tables, including detection limits and detection frequencies.

Field replicates were averaged prior to statistical analysis.

[a] Results of statistical comparison. Bold indicates chemicals for which statistical testing showed site concentrations to be greater than reference concentrations at a significance level of alpha = 0.05.

[b] Confidence interval for the site mean straddles zero, due to small sample size and/or high variability.

Table 3-10. Statistical comparison of site and reference lagoon sediment data

	Reference					Site					Site >
Chemical	N	Min	Max	Mean	Stdev	N	Min	Max	Mean	Stdev	Reference?[a]
Aluminum	3	7,440	14,800	11,147	3,680	8	2,450	14,300	7,574	4,548	no
Antimony	3	0.010	0.12	0.08	0.059	8	0.070	0.27	0.16	0.0725	no
Arsenic	3	2.6	4.9	4.0	1.2	8	5.3	17.9	7.8	4.2	**yes**
Barium	3	164	271	226	55.5	8	54.1	350	234	97.8	no
Cadmium	13										>50% ND
Chromium	3	12.5	24.9	19.6	6.4	8	4.1	27.2	13.8	8.6	no
Cobalt	3	5.0	9.7	6.8	2.5	8	3.9	11.8	7.1	2.8	no
Copper	3	9.9	18.7	14.7	4.5	8	3.0	28.2	14.2	8.6	no
Fluoride	3										>50% ND
Iron	3	14,000	22,200	19,233	4,546	8	10,100	75,000	27,150	21,985	no
Lead	28	2.4	31.0	11.1	6.6	26	4.7	302	44.4	68.8	**yes**
Manganese	3	75.5	129	99.9	27.1	8	97.9	274	158	55.899	**yes**
Mercury	3	0.030	0.060	0.050	0.017	8	0.0040	0.096	0.0486	0.0316	no
Molybdenum	3	0.46	0.98	0.77	0.28	8	0.41	3.4	1.4	1.2	no
Nickel	3	18.7	37.0	27.2	9.2	8	12.0	39.0	24.2	10.0	no
Selenium	3	0.60	1.4	1.1	0.44	8	0.10	2.2	1.0	0.69	no
Silver	3	0.010	0.11	0.067	0.051	8	0.020	0.27	0.12	0.0828	no
Strontium	3	20.9	40.0	31.9	9.9	8	10.4	108	51.8	32.6	no
Thallium	3	0.038	0.10	0.081	0.037	8	0.018	0.18	0.0754	0.0558	no
Tin	3										>50% ND
Vanadium	3	16.8	31.5	25.2	7.6	8	8.5	35.1	21.7	10.7	no
Zinc	28	16.0	371	92.7	63.8	26	36.0	1590	242	319	**yes**

Note: CoPC - chemical of potential concern
ND - not detected

Concentrations are given in mg/kg dry weight.

Undetected values are treated as detections at one-half the detection limit for the statistical analysis. In cases where greater than or equal to 50 percent of the site values were undetected, statistical analyses were not performed. Further summary information is provided in the CoPC screening tables, including detection limits and detection frequencies.

Field replicates were averaged prior to statistical analysis.

[a] Results of statistical comparison. Bold indicates chemicals for which statistical testing showed site concentrations to be greater than reference concentrations at a significance level of alpha = 0.05.

Table 3-11. Statistical comparison of site and reference lagoon surface water data

	Reference					Site					Site >
Chemical	N	Min	Max	Mean	Stdev	N	Min	Max	Mean	Stdev	Reference?[a]
Aluminum	3	53.5	434	182	218	8	19.7	247	81.6	77.6	no
Antimony	3	0.11	0.13	0.12	0.010	8	0.19	0.63	0.32	0.15	**yes**
Arsenic	3	52.9	98.8	76.3	23.0	8	4.5	126	56.2	48.4	no
Barium	3	144	168	156	12.0	8	112	413	233	118	no
Cadmium	5	0.050	0.26	0.15	0.10	14	0.040	0.30	0.13	0.10	no
Chromium	3	6.0	8.2	7.2	1.1	8	1.7	4.5	2.7	1.1	no
Cobalt	3	3.7	5.4	4.4	0.86	8	0.45	1.4	0.90	0.35	no
Copper	3	0.40	1.4	0.80	0.53	8	0.50	1.4	0.96	0.29	no
Fluoride	3	10.0	20.0	13.3	5.77	8	50.0	200	114	73.3	**yes**
Iron	3	290	693	427	230	8	200	723	445	211	no
Lead	5	0.095	0.85	0.29	0.32	14	0.40	2.3	1.0	0.72	**yes**
Manganese	3	492	801	598	176	8	13.9	277	84.5	98.3	no
Mercury											>50% ND
Molybdenum	3	0.070	0.090	0.080	0.010	8	0.30	2.4	1.2	0.85	**yes**
Nickel	3	9.2	15.2	11.5	3.2	8	3.5	10.6	7.1	2.7	no
Selenium	3	0.20	0.20	0.20	0	8	0.20	0.60	0.30	0.13	no
Silver	3	0.020	0.030	0.023	0.0058	8	0.010	0.25	0.11	0.09	no[b]
Strontium	3	991	1,470	1,157	271	8	505	1,850	1,226	546	no
Thallium	3	0.0060	0.0090	0.0080	0.0017	8	0.0070	0.070	0.029	0.021	no
Tin											>50% ND
Vanadium	3	0.40	0.40	0.40	0	8	0.18	0.85	0.42	0.21	no
Zinc	5	17.0	30.1	21.3	5.3	14	3.1	110	25.1	32.6	no[b]

Note: CoPC - chemical of potential concern
ND - not detected

Concentrations are given in µg/L unfiltered, except for cadmium, lead, and zinc, which are µg/L dissolved.

Undetected values are treated as detections at one-half the detection limit for the statistical analysis. In cases where greater than or equal to 50 percent of the site values were undetected, statistical analyses were not performed. Further summary information is provided in the CoPC screening tables, including detection limits and detection frequencies.

Field replicates were averaged prior to statistical analysis.

[a] Results of statistical comparison. Bold indicates chemicals for which statistical testing showed site concentrations to be greater than reference concentrations at a significance level of alpha = 0.05.

[b] Confidence interval for the site mean straddles zero, due to small sample size and/or high variability.

Table 3-12. Statistical comparison of site and reference marine sediment data

	Reference					Site					Site >
Chemical	N	Min	Max	Mean	Stdev	N	Min	Max	Mean	Stdev	Reference?[a]
Aluminum	15	1,970	8,000	5,043	1,731	18	1,990	6,070	4,700	1,164	no
Antimony											>50% ND
Arsenic	21	5.6	13.0	8.7	1.9	69	3.1	14.5	7.3	1.8	R>S
Barium	21	22.0	431	207	100	69	79.5	639	239	98.0	**yes**
Cadmium	21	0.020	0.23	0.068	0.044	129	0.020	52.9	1.0	4.7	no[b]
Chromium	21	1.4	18.0	11.9	4.4	69	2.4	33.5	14.6	5.7	no
Cobalt	9	4.2	8.7	7.2	1.7	18	3.2	8.9	6.8	1.3	no
Copper	21	3.0	10.2	6.4	1.7	69	3.7	34.8	7.8	4.4	no
Fluoride	9	0.40	2.0	1.3	0.51	16	0.40	1.5	1.1	0.29	no
Iron	15	8,150	22,700	15,149	4,973	18	9,960	19,300	15,987	2,570	no
Lead	21	2.7	11.2	5.3	1.7	129	1.6	5,620	58.5	494	no[b]
Manganese	9	187	389	301	71.7	18	161	363	276	59.6	no
Mercury											>50% ND
Molybdenum	9	0.44	0.83	0.57	0.12	18	0.37	1.4	0.58	0.26	no
Nickel	15	9.8	34.8	22.5	7.2	18	11.3	33.3	24.3	6.3	no
Selenium											>50% ND
Silver	21	0.020	0.49	0.12	0.17	69	0.030	2.1	0.50	0.61	**yes**
Strontium	9	13.0	29.0	24.0	5.5	17	24.4	33.8	28.0	2.5	**yes**
Thallium	9	0.025	0.052	0.037	0.0096	17	0.026	1.1	0.098	0.27	no
Tin											>50% ND
Vanadium	9	13.0	33.9	22.2	6.6	41	9.1	46.0	27.9	9.7	no
Zinc	21	25.0	56.8	42.2	8.5	129	5.5	2,550	87.3	237	**yes**

Note: CoPC - chemical of potential concern
ND - not detected

Concentrations are given in mg/kg dry weight.

Undetected values are treated as detections at one-half the detection limit for the statistical analysis. In cases where greater than or equal to 50 percent of the site values were undetected, statistical analyses were not performed. Further summary information is provided in the CoPC screening tables, including detection limits and detection frequencies.

Field replicates were averaged prior to statistical analysis.

[a] Results of statistical comparison. Bold indicates chemicals for which statistical testing showed site concentrations to be greater than reference concentrations at a significance level of alpha = 0.05.

[b] Confidence interval for the site mean straddles zero, due to small sample size and/or high variability.

Table 3-13. Statistical comparison of site and reference marine surface water data

	Reference					Site					Site >
Chemical	N	Min	Max	Mean	Stdev	N	Min	Max	Mean	Stdev	Reference?[a]
Aluminum	6	25.0	336	170	148	9	25.0	205	102	69.7	no
Antimony	6	0.20	1.7	0.70	0.62	9	0.20	1.9	0.64	0.55	no
Arsenic	6	1.1	7.5	3.8	2.9	9	1.5	6.0	3.2	1.7	no
Barium	6	9.9	38.1	23.2	14.1	9	12.1	39.4	21.3	10.6	no
Cadmium	6	2.3	4.7	3.5	1.2	9	1.6	4.6	2.9	1.3	no
Chromium											100% ND
Cobalt	6	4.0	4.5	4.3	0.17	9	3.9	4.6	4.2	0.22	no
Copper											>50% ND
Fluoride	6	600	800	700	110	9	500	900	733	158	no
Iron	6	33.6	643	314	306	9	52.3	375	171	133	no
Lead	6	0.76	1.3	0.99	0.18	9	0.80	1.3	1.0	0.21	no
Manganese	6	10.1	25.5	17.0	7.3	9	13.1	31.9	19.2	5.9	no
Mercury											100% ND
Molybdenum	6	8.3	10.6	9.5	0.93	9	8.4	11.0	9.9	1.1	no
Nickel											100% ND
Selenium	6	0.20	0.50	0.28	0.12	9	0.20	1.0	0.49	0.24	**yes**
Silver	6	0.10	0.27	0.18	0.066	9	0.10	0.95	0.40	0.28	no
Strontium	6	4,530	5,290	4,900	369	9	4,420	5,600	5,128	444	no
Thallium											>50% ND
Tin											>50% ND
Vanadium											>50% ND
Zinc											100% ND

Note: CoPC - chemical of potential concern
ND - not detected

Concentrations are given in μg/L unfiltered.

Undetected values are treated as detections at one-half the detection limit for the statistical analysis. In cases where greater than or equal to 50 percent of the site values were undetected, statistical analyses were not performed. Further summary information is provided in the CoPC screening tables, including detection limits and detection frequencies.

Field replicates were averaged prior to statistical analysis.

[a] Results of statistical comparison. Bold indicates chemicals for which statistical testing showed site concentrations to be greater than reference concentrations at a significance level of alpha = 0.05.

Table 3-14. Human health chemical of potential concern screening results for surface soil

Scenario Timeframe: Current/Future
Medium: Soil
Exposure Medium: Surface Soil

Exposure Point	Chemical	Minimum Detected Concentration	Maximum Detected Concentration	Units	Location of Maximum Concentration	Detection Frequency	Range of Detection Limits	Concentration Used for Screening[a]	Reference Range[b]	Residential Screening Toxicity Value[c]		Frequency of Detected Values Exceeding Criteria	Frequency of Reference Values Exceeding Criteria	Non-Residential Screening Toxicity Value[d]		CoPC Flag (Y/N)	Rationale for Selection or Deletion
All Site Surface Soil																	
	Aluminum	1,180	16,600	mg/kg	RF-05	51/51	--	16,600	1,640–12,400	13,688	N	2/51	0/10	255,500	N	No	REF
	Antimony	0.38 *J*	14.8	mg/kg	CAG-W29	13/40	5–26	14.8	0.17–0.6	5.5	N	1/40	0/5	102	N	No	IFS
	Arsenic	1.3	93.6	mg/kg	CAG-H30	54/75	10–51	93.6	4.15–35	0.8	C	54/75	10/10	77	C	No	REF
	Barium	357	7,090	mg/kg	RF-07	40/40	--	7,090	109–622	960	N	35/40	0/5	17,885	N	Yes	ASL
	Cadmium	1.0	388 *J*	mg/kg	CAG-H30	430/478	0.4–2.5	388	0.24–3.59	14	N	236/478	0/10	256	N	Yes	ASL
	Chromium	4.86	24	mg/kg	RF-05	40/40	--	24	4.94–19.3	41	N	0/40	0/5	767	N	No	REF/BSL
	Cobalt	4.21	27	mg/kg	RF-05	39/40	5–5	27	7.28–20.6	274	N	0/40	0/5	5,110	N	No	REF/BSL
	Copper	9.76	109	mg/kg	RAT5-0NA	40/40	--	109	14.3–46.5	548	N	0/40	0/5	10,220	N	No	REF/BSL
	Fluoride	0.5 *J*	1.3 *J*	mg/kg	RF-16	9/12	0.4–0.4	1.3	0.3–0.5	821	N	0/12	0/5	15,330	N	No	BSL
	Iron	2,650	35,000	mg/kg	CAG-W29	51/51	--	35,000	5,750–72,600	4,106	N	49/51	10/10	76,650	N	No	REF
	Lead	13.5	48,300	mg/kg	1007468	467/479	8.5–12	48,300	8.75–142	400	N	279/479	0/10	1,000	N	Yes	ASL
	Manganese	280	1,000	mg/kg	170_C1	40/40	--	1,000	250–4,080	329	N	37/40	4/5	6,132	N	No	REF
	Mercury	0.1	1.69	mg/kg	RF-107	12/12	--	1.69	0.05–0.18	2.6	N	0/12	0/5	77	N	No	BSL
	Molybdenum	0.35	3.3	mg/kg	RF-07	16/40	0.9–5.1	3.3	0.27–2.8	68	N	0/40	0/5	1,278	N	No	BSL
	Nickel	17.3	56.8	mg/kg	RC-06-A	40/40	--	56.8	23.5–51.4	270	N	0/40	0/5	5,110	N	No	REF/BSL
	Selenium	0.3 *J*	3 *J*	mg/kg	RF-107	12/30	10–51	3	0.5–1	68	N	0/30	0/5	1,278	N	No	BSL
	Silver	0.14	8.3	mg/kg	RAT5-0NA	21/40	0.9–5.1	8.3	0.05–0.25	68	N	0/40	0/5	1,278	N	No	BSL
	Strontium	36.2	90.1	mg/kg	RF-16	20/20	--	90.1	9.3–63.6	8,213	N	0/20	0/5	153,300	N	No	BSL
	Thallium	0.112	1.32	mg/kg	RF-32	12/12	--	1.32	0.1–0.24	0.9	N	1/12	0/5	17	N	No	IFS
	Tin	3.9 *J*	6 *J*	mg/kg	RF-27	2/27	2.25–26	6	ND	8,213	N	0/27	0/5	153,300	N	No	BSL
	Vanadium	7.94	31.8	mg/kg	RF-05	40/40	--	31.8	5.62–19.2	96	N	0/40	0/5	1,789	N	No	REF/BSL
	Zinc	37.4	64,300	mg/kg	CAG-H30	479/479	--	64,300	72.5–753	4,100	N	158/479	0/10	76,650	N	Yes	ASL

Note: All results reported as dry weight.
For the purposes of screening, field replicates have been averaged.
-- - not applicable
C - carcinogenic based on a cancer risk of 1×10^{-6}
CoPC - chemical of potential concern
J - estimated value
N - noncarcinogenic based on hazard quotient of 0.1
ND - not detected

Rationale Codes:
Selection Reason:
ASL - above screening levels

Deletion Reason:
BSL - below screening level
IFS - infrequently above screening level
REF - below or consistent with reference levels

[a] The maximum detected soil concentration was used for screening CoPCs.

[b] The reference range corresponds to road material site soil samples from areas unimpacted by fugitive dust.

[c] Residential screening toxicity values represent arctic zone soil cleanup levels (from 18 AAC 75.341, Table B1) divided by 10. Where no Table B1 value exists, screening values were calculated based on residential formulas and input parameters provided in DEC (2002).

[d] Non-residential screening toxicity values using industrial formulas and input parameters provided in DEC (2000).

Table 3-15. Human health screening results for drinking water ingestion in stream surface water

Scenario Timeframe: Current/Future
Medium: Water
Exposure Medium: Stream Surface Water for Drinking Water Ingestion

Exposure Point	Chemical	Minimum Detected Concentration	Maximum Detected Concentration	Units	Location of Maximum Concentration	Detection Frequency	Range of Detection Limits	Concentration Used for Screening[a]	Reference Range[b]	Screening Toxicity Value[c]		Potential ARAR/TBC Value	Potential C Source[d]	Frequency of Detected Values Exceeding Criteria	Frequency of Reference Values Exceeding Criteria	CoPC Flag (Y/N)	Rationale for Selection or Deletion
All Site Stream Surface Water																	
	Aluminum	6.45	4,060	µg/L	StrRd	133/230	2.52–10	4,060	17.3–2,770	3,650	N	50–200	MCL	2/230	0/3	No	IFS/REF
	Antimony	0.14	0.6	µg/L	NHDowRd	6/14	0.063	0.6	ND–0.08	0.6	N	6	MCL	1/14	0/3	No	BSL
	Arsenic	ND	ND	µg/L	--	0/14	0.482	ND	ND–2.2	5	C	50	MCL	ND	0/3	No	IFD/BSL
	Barium	12.2	266	µg/L	NHNFUp	14/14	--	266	86.1–222	200	N	2,000	MCL	1/14	1/3	No	REF/IFS
	Cadmium	0.03	0.40	µg/L	Various	24/229	0.02–0.25	0.4	0.01–0.07	0.5	N	5	MCL	0/229	0/3	No	BSL
	Chromium	ND	ND	µg/L	--	0/18	0.4	ND	0.17–3.71	10	N	100	MCL	ND	0/3	No	IFD/BSL
	Cobalt	0.03	0.33	µg/L	NHRoad	12/14	0.01	0.33	0.12–2.72	73	N	--	--	0/14	0/3	No	BSL/REF
	Copper	0.3	1.2	µg/L	OmiDowRd	16/18	0.11	1.2	0.6–5.4	130	N	1,300	MCL	0/18	0/3	No	BSL/REF
	Fluoride	40	120	µg/L	NHRoad	27/31	50	120	30–40	219,000	N	--	--	0/31	0/3	No	BSL
	Iron	6	10,300	µg/L	StrRd	186/230	2.57–25	10,300	64.2–6,710	1,095	N	300	MCL	11/230	1/3	No	IFS/REF
	Lead	0.0	7.34	µg/L	StrDowRd	84/230	0.02–0.401	7.34	0.02–1.91	1.5	N	15	MCL	5/230	1/3	No	IFS
	Manganese	0.56	36	µg/L	MudLkCr	18/18	--	36	4.87–128	87.6	N	50	MCL	0/18	1/3	No	BSL/REF
	Mercury	ND	ND	µg/L	--	0/14	0.0179	ND	ND	0.2	N	2	MCL	ND	ND	No	IFD/BSL
	Molybdenum	0.37	2.27	µg/L	NHDowRd	11/14	0.178	2.27	0.05–0.17	18.25	N	--	--	0/14	0/3	No	BSL
	Nickel	0.26	6.71	µg/L	NHRoad	14/14	--	6.71	1.06–10.5	10	N	100	MCL	0/14	1/3	No	BSL/REF
	Selenium	0.067	1.24	µg/L	TutMth	15/29	0.0201	1.24	ND	5	N	50	MCL	0/29	ND	No	BSL
	Silver	ND	ND	µg/L	--	0/14	0.023	ND	ND–0.03	18	N	100	MCL	ND	0/3	No	IFD/BSL
	Strontium	19.4	172	µg/L	NHDowRd	14/14	--	172	32.5–81.1	2,190	N	--	--	0/14	0/3	No	BSL/REF
	Thallium	0.04	0.55	µg/L	AufRd	9/29	0.0155–0.07	0.55	ND–0.014	0.2	N	2	MCL	2/29	0/3	No	IFS
	Tin	1.3	5.33	µg/L	OmiNFUp	5/14	0.59	5.33	ND	2,190	N	--	--	0/14	ND	No	BSL
	Vanadium	0.67	0.93	µg/L	ARC-U	4/14	0.335	0.93	0.16–5.57	26	N	--	--	0/14	0/3	No	BSL/REF[e]
	Zinc	1.0	60.1	µg/L	TutDowRd	107/230	0.5–5	60	0.31–9.84	1,100	N	5,000	MCL	0/230	0/3	No	BSL

Note: All results reported as unfiltered.
For the purposes of screening, field replicates have been averaged.

-- - not applicable
ARAR - applicable or relevant and appropriate requirement
C - carcinogenic based on a cancer risk of 1×10^{-6}
CoPC - chemical of potential concern
J - estimated value
MCL - maximum contaminant level
N - noncarcinogenic based on hazard quotient of 0.1
ND - not detected
TBC - to be continued

Rationale Codes:

Selection Reason:
ASL - above screening levels

Deletion Reason:
BSL - below screening level
IFD - infrequently, or not detected
IFS - infrequently above screening level
REF - below or consistent with reference levels

[a] The maximum detected stream surface water concentration was used for screening CoPCs.

[b] The reference range corresponds to stream surface water samples taken from areas unimpacted by fugitive dust.

[c] Screening toxicity values represent arctic zone drinking water cleanup levels (from 18 AAC 75.345, Table C) divided by 10. Where no Table C value exists, screening values were calculated based on residential drinking water formulas and input parameters provided in DEC (2002).

[d] An ARAR listed as an MCL is a maximum contaminant level derived by EPA, and is considered protective of the water body for use as the sole domestic drinking water source.

[e] There was an inadequate number of detections to statistically test for a difference between site and reference concentrations; however, the analyte was screened out because the maximum site concentration was less than the maximum reference concentration.

Table 3-16. Human health screening results for fish consumption in stream surface water

Scenario Timeframe: Current/Future
Medium: Water
Exposure Medium: Stream Surface Water for Fish Consumption

Exposure Point	Chemical	Minimum Detected Concentration	Maximum Detected Concentration	Units	Location of Maximum Concentration	Detection Frequency	Range of Detection Limits	Concentration Used for Screening[a]	Reference Range[b]	Screening Toxicity Value[c]		Potential ARAR/TBC Value	Potential ARAR/TBC Source[d]	Frequency of Detected Values Exceeding Criteria	Frequency of Reference Values Exceeding Criteria	CoPC Flag (Y/N)	Rationale for Selection or Deletion
All Site Stream Surface Water																	
	Aluminum	6.45	4,060	µg/L	StrRd	133/230	2.52–10	4,060	17.3–2,770	--		--	--	--	--	No	REF
	Antimony	0.14	0.6	µg/L	NHDowRd	6/14	0.063	0.6	ND–0.08	5.6	N	--	--	0/14	0/3	No	BSL
	Arsenic	ND	ND	µg/L	--	0/14	0.482	ND	ND–2.2	0.018	C	0.00982	WDOE	ND	1/3	No	IFD/BSL
	Barium	12.2	266	µg/L	NHNFUp	14/14	--	266	86.1–222	--		--	--	--	--	No	REF
	Cadmium	0.03	0.40	µg/L	Various	24/229	0.02–0.25	0.4	0.01–0.07	--		5.06	WDOE	--	--	No	BWC
	Chromium	ND	ND	µg/L	--	0/18	0.4	ND	0.17–3.71	--		203	WDOE	--	--	No	IFD/BWC
	Cobalt	0.03	0.33	µg/L	NHRoad	12/14	0.01	0.33	0.12–2.72	--		--	--	--	--	No	REF
	Copper	0.3	1.2	µg/L	OmiDowRd	16/18	0.11	1.2	0.6–5.4	1,300	N	2,660	WDOE	0/18	0/3	No	BSL/REF
	Fluoride	40	120	µg/L	NHRoad	27/31	50	120	30–40	--		--	--	--	--	No	NSC
	Iron	6	10,300	µg/L	StrRd	186/230	2.57–25	10,300	64.2–6,710	--		--	--	--	--	No	REF
	Lead	0.0	7.34	µg/L	StrDowRd	84/230	0.02–0.401	7.34	0.02–1.91	--		--	--	--	--	No	NSC
	Manganese	0.56	36	µg/L	MudLkCr	18/18	--	36	4.87–128	--		--	--	--	--	No	REF
	Mercury	ND	ND	µg/L	--	0/14	0.0179	ND	ND	--		--	--	--	--	No	IFD
	Molybdenum	0.37	2.27	µg/L	NHDowRd	11/14	0.178	2.27	0.05–0.17	--		--	--	--	--	No	NSC
	Nickel	0.26	6.71	µg/L	NHRoad	14/14	--	6.71	1.06–10.5	610	N	1,100	WDOE	0/14	1/3	No	BSL/REF
	Selenium	0.067	1.24	µg/L	TutMth	15/29	0.0201	1.24	ND	170	N	--	--	0/29	ND	No	BSL
	Silver	ND	ND	µg/L	--	0/14	0.023	ND	ND–0.03	--		6,480	WDOE	--	--	No	IFD/BWC
	Strontium	19.4	172	µg/L	NHDowRd	14/14	--	172	32.5–81.1	--		--	--	--	--	No	REF
	Thallium	0.04	0.55	µg/L	AufRd	9/29	0.0155–0.07	0.55	ND–0.014	1.7	N	1.56	WDOE	0/29	0/3	No	BSL
	Tin	1.3	5.33	µg/L	OmiNFUp	5/14	0.59	5.33	ND	--		--	--	--	--	No	NSC
	Vanadium	0.67	0.93	µg/L	ARC-U	4/14	0.335	0.93	0.16–5.57	--		--	--	--	--	No	REF[e]
	Zinc	1.0	60.1	µg/L	TutDowRd	107/230	0.5–5	60	0.31–9.84	7,400	N	16,500	WDOE	0/230	0/3	No	BSL

Note: All results reported as unfiltered.
For the purposes of screening, field replicates have been averaged.

-- - not applicable
ARAR - applicable or relevant and appropriate requirement
AWQC - ambient water quality criteria
C - carcinogenic based on a cancer risk of 1×10^{-6}
CoPC - chemical of potential concern
J - estimated value
MTCA - Model Toxics Control Act
N - noncarcinogenic based on hazard quotient of 0.1
ND - not detected
TBC - to be considered
WDOE - Washington State Department of Ecology

Rationale Codes:

Selection Reason:
ASL - above screening levels

Deletion Reason:
BSL - below screening level
BWC - no AWQC available, but below MTCA surface water criteria for bioaccumulation in fish
IFD - infrequently, or not detected
IFS - infrequently above screening level
NSC - no screening criteria
REF - below or consistent with reference levels

[a] The maximum detected stream surface water concentration was used for screening CoPCs.

[b] The reference range corresponds to stream surface water samples taken from areas unimpacted by fugitive dust.

[c] Screening toxicity values represent the AWQC protective for human consumption of seafood and domestic drinking water usage from the water body (U.S. EPA 2002c). The AWQC were modified, when necessary, to assume a target hazard quotient of 0.1 and a target risk of 10^{-6}. The ARAR represents the Washington State cleanup level for surface water and is protective of bioaccumulation into, and human consumption of, seafood (WDOE 1996).

[d] The ARARs represent the Washington State Department of Ecology cleanup level for surface water and is protective of bioaccumulation into, and human consumption of, fish (WDOE 1996).

[e] There was an inadequate number of detections to statistically test for a difference between site and reference concentrations; however, the analyte was screened out because the maximum site concentration was less than the maximum reference concentration.

Table 3-17. Human health screening results for lagoon water

Scenario Timeframe: Current/Future
Medium: Lagoon Water
Exposure Medium: Lagoon Water

Exposure Point	Chemical	Minimum Detected Concentration	Maximum Detected Concentration	Units	Location of Maximum Concentration	Detection Frequency	Range of Detection Limits	Concentration Used for Screening[a]	Reference Range[b]	Screening Toxicity Value[c]	Potential ARAR/TBC Value	Potential ARAR/TBC Source[d]	Frequency of Detected Values Exceeding Criteria	Frequency of Reference Values Exceeding Criteria	CoPC Flag (Y/N)	Rationale for Selection or Deletion
Lagoon Water																
	Aluminum	19.7	247	µg/L	IP-04	8/8		247	53.5–434	--	--	--	--	--	No	REF
	Antimony	0.19	0.63	µg/L	PLNL	8/8		0.63	0.11–0.13	64 N	--	--	0/8	0/3	No	BSL
	Arsenic	4.5	126	µg/L	IP-04	8/8		126	52.9–98.8	0.14 C	0.00982	WDOE	8/8	3/3	No	REF
	Barium	112	413	µg/L	PLNL	8/8		413	144–168	--	--	--	--	--	No	REF
	Cadmium	0.04 *J*	0.3	µg/L	NLH	11/14	0.1	0.3	ND–0.26	--	5.06	WDOE	--	--	No	REF/BWC
	Chromium	1.69	4.49	µg/L	IP-04	8/8		4.49	5.96–8.22	--	203	WDOE	--	--	No	REF/BWC
	Cobalt	0.45	1.38	µg/L	PLNL	8/8		1.38	3.7–5.35	--	--	--	--	--	No	REF
	Copper	0.5 *J*	1.4	µg/L	IP-03	8/8		1.4	0.4–1.4	--	2,660	WDOE	--	--	No	REF/BWC
	Fluoride	50 *J*	200	µg/L	IP-01,IP-02,IP-04	8/8		200	ND–20	--	--	--	--	--	No	NSC
	Iron	200	723	µg/L	PLNN	8/8		723	290–693	--	--	--	--	--	No	REF
	Lead	0.4	2.3	µg/L	PLNP	14/14		2.3	ND–5.9	--	--	--	--	--	No	NSC
	Manganese	13.9	277	µg/L	PLNN	8/8		277	492–801	--	--	--	--	--	No	REF
	Mercury	ND	ND	µg/L	ND	0/8		ND	ND	--	--	--	--	--	No	IFD
	Molybdenum	0.3	2.41	µg/L	IP-04	8/8	0.1	2.41	0.07–0.09	--	--	--	--	--	No	NSC
	Nickel	3.5	10.6	µg/L	IP-01	8/8		10.6	9.9–15.2	460 N	1,100	WDOE	0/8	0/3	No	REF/BSL
	Selenium	0.3 *J*	0.6 *J*	µg/L	PLNN	5/8	0.4	0.6	ND	420 N	--	--	0/8	0/3	No	REF/BSL
	Silver	0.01 *J*	0.25	µg/L	PLNL	7/8	0.1	0.25	0.02–0.03	--	6,480	WDOE	--	--	No	REF/BWC
	Strontium	505	1,850	µg/L	PLNN	8/8		1,850	991–1,470	--	--	--	--	--	No	REF
	Thallium	0.007 *J*	0.07 *J*	µg/L	NLF,PLNL	4/8	0.026-0.06	0.07	0.006–0.009	0.63 N	1.56	WDOE	0/8	0/3	No	REF/BSL
	Tin	23.7 *J*	23.7 *J*	µg/L	NLF	1/8	20	23.7	ND	--	--	--	--	--	No	NSC
	Vanadium	0.22	0.85 *J*	µg/L	IP-04	5/8	0.36-0.8	0.85	ND	--	--	--	--	--	No	REF
	Zinc	3.09 *J*	110	µg/L	NLH	14/14		110	ND–30.1	2,600 N	16,500	WDOE	0/14	0/3	No	REF/BSL

Note: All results reported as unfiltered.
For the purposes of screening, field replicates have been averaged.
-- - not applicable
ARAR - applicable or relevant and appropriate requirement
AWQC - ambient water quality criteria
C - carcinogenic based on a cancer risk of 1×10^{-b}
CoPC - chemical of potential concern
J - estimated value
MTCA - Model Toxics Control Act
N - noncarcinogenic based on hazard quotient of 0.1
ND - not detected
TBC - to be considered
WDOE - Washington State Department of Ecology

Rationale Codes:
Selection Reason:
ASL - above screening levels

Deletion Reason:
BSL - below screening level
BWC - no AWQC available, but below MTCA surface water criteria for bioaccumulation in fish
IFD - infrequently, or not detected
NSC - no screening criteria
REF - below or consistent with reference levels

[a] The maximum detected lagoon surface water concentration was used for screening CoPCs.

[b] The reference range corresponds to lagoon surface water samples taken from areas unimpacted by fugitive dust.

[c] Screening toxicity values represent the AWQC protective for human consumption of seafood from the water body (U.S. EPA 2002c). The AWQC were modified, when necessary, to assume a target hazard quotient of 0.1 and a target risk of 10^{-6}.

[d] The ARAR represents the Washington State cleanup level for surface water and is protective of bioaccumulation into, and human consumption of, seafood (WDOE 1996).

Table 3-18. Human health screening results for marine surface water

Scenario Timeframe: Current/Future
Medium: Marine Water
Exposure Medium: Marine Surface Water

Exposure Point	Chemical	Minimum Detected Concentration	Maximum Detected Concentration	Units	Location of Maximum Concentration	Detection Frequency	Range of Detection Limits	Concentration Used for Screening[a]	Reference Range[b]	Screening Toxicity Value[c]	Potential ARAR/TBC Value	Potential ARAR/TBC Source[d]	Frequency of Detected Values Exceeding Criteria	Frequency of Reference Values Exceeding Criteria	CoPC Flag (Y/N)	Rationale for Selection or Deletion
Marine Water																
	Aluminum	43	205	µg/L	NML	8/9	25	205	ND–336	--	--	--	--	--	No	REF
	Antimony	0.30 *J*	1.88 *J*	µg/L	NML	5/9	0.2–0.4	1.88	ND–1.67	64 N	--	--	0/9	0/6	No	REF/BSL
	Arsenic	1.5 *J*	6.0 *J*	µg/L	NMAA	7/9	3	6	ND–7.5	0.14 C	0.00982	WDOE	7/9	5/6	No	REF
	Barium	12.1	39.4	µg/L	NMK	9/9	--	39.4	9.91–38.1	--	--	--	--	--	No	REF
	Cadmium	1.6	4.6	µg/L	NML	9/9	--	4.6	2.27–4.69	--	5.06	WDOE	--	--	No	REF/BWC
	Chromium	ND	ND	µg/L	--	0/9	1–2	ND	ND	--	203	WDOE	--	--	No	IFD/BSL
	Cobalt	3.85	4.60	µg/L	NMG	9/9	--	4.6	4.03–4.48	--	--	--	--	--	No	REF
	Copper	1.0	3.6	µg/L	NML	4/9	4	3.6	ND–2.6	--	2,660	WDOE	--	--	No	BWC
	Fluoride	500	900	µg/L	NMAA, NMG, NML	9/9	--	900	600–800	--	--	--	--	--	No	REF
	Iron	52	375	µg/L	NMAA	9/9	--	375	33.6–643	--	--	--	--	--	No	REF
	Lead	0.8 *J*	1.34	µg/L	NMAA	9/9	--	1.34	0.76–1.25	--	--	--	--	--	No	REF
	Manganese	13.1	31.9	µg/L	NMK	9/9	--	31.9	10.1–25.5	--	--	--	--	--	No	REF
	Mercury	ND	ND	µg/L	--	0/9	0.05	ND	ND	--	--	--	--	--	No	IFD
	Molybdenum	8.4	11	µg/L	NMAA, NMG, NML	9/9	--	11	8.26–10.6	--	--	--	--	--	No	REF
	Nickel	ND	ND	µg/L	--	0/9	2-10	ND	ND	460 N	1,100	WDOE	0/9	0/6	No	IFD/BSL
	Selenium	0.3 *J*	1 *J*	µg/L	NMG	8/9	0.2	1	ND–0.5	420 N	--	--	0/9	0/6	No	BSL
	Silver	0.4	0.95	µg/L	NMAA	5/9	0.1–0.2	0.95	ND–0.27	--	6,480	WDOE	--	--	No	REF/BWC
	Strontium	4,420	5,600 *J*	µg/L	NMG, NML	9/9	--	5,600	4,530–5,290	--	--	--	--	--	No	REF
	Thallium	0.09 *J*	0.09 *J*	µg/L	NMAA	1/9	0.05–0.1	0.09	ND–0.133	0.63 N	1.56	WDOE	0/9	0/6	No	BSL
	Tin	23.3 *J*	23.3 *J*	µg/L	NMAA	1/9	3–10	23.3	ND–26.4	--	--	--	--	--	No	REF[e]
	Vanadium	4.44 *J*	5.27 *J*	µg/L	NMK	2/9	1.4–2	5.27	ND–8.44	--	--	--	--	--	No	REF[e]
	Zinc	ND	ND	µg/L	--	0/9	1	ND	ND	2,600 N	16,500	WDOE	0/9	0/6	No	IFD/BSL

Note: All results reported as unfiltered.
For the purposes of screening, field replicates have been averaged.
-- - not applicable
ARAR - applicable or relevant and appropriate requirement
AWQC - ambient water quality criteria
C - carcinogenic based on a cancer risk of 1×10^{-6}
CoPC - chemical of potential concern
J - estimated value
MTCA - Model Toxics Control Act
N - noncarcinogenic based on hazard quotient of 0.1
ND - not detected
TBC - to be considered
WDOE - Washington State Department of Ecology

Rationale Codes:
Selection Reason:
ASL - above screening levels

Deletion Reason:
BSL - below screening level
BWC - no AWQC available, but below MTCA surface water criteria for bioaccumulation in fish
IFD - infrequently, or not detected
REF - below or consistent with reference levels

[a] The maximum detected marine surface water concentration was used for screening CoPCs.

[b] The reference range corresponds to marine surface water samples taken from areas unimpacted by fugitive dust.

[c] Screening toxicity values represent the AWQC protective for human consumption of seafood from the water body (U.S. EPA 2002c). The AWQC were modified, when necessary, to assume a target hazard quotient of 0.1 and a target risk of 10^{-6}.

[d] The ARAR represents the Washington State cleanup level for surface water and is protective of bioaccumulation into, and human consumption of, seafood (WDOE 1996).

[e] There were an inadequate number of detections to statistically test for a difference between site and reference concentrations; however, the analyte was screened out because the maximum site concentration was less than the maxi reference concentration.

Table 3-19. Ecological screening results for tundra soil

Analyte	Detection Frequency	Minimum Detected Concentration (mg/kg)	Maximum Detected Concentration (mg/kg)	Detection Limits (mg/kg)	Reference Concentration		Ecological Screening Benchmark			Detection Frequency Above Benchmark[a]		
					Minimum Detected Concentration (mg/kg)	Maximum Detected Concentration (mg/kg)	ORNL Terrestrial Plants[b] (mg/kg)	ORNL Earthworms[c] (mg/kg)	ORNL Soil Microorganisms[c] (mg/kg)	ORNL Terrestrial Plants	ORNL Earthworms	ORNL Soil Microorganisms
Aluminum	32/32	358	18,900	-	368	11,300	50	--	600	32/32	--	29/32
Antimony	14/26	0.15	25.8	7-29	0.11	0.28	5	--	--	3/26	--	--
Arsenic	17/32	0.3	150	19–58	0.4	6.8	10	60	100	6/32	1/32	1/32
Barium	26/26	53	5,810	-	108	624	500	--	3,000	13/26	--	3/26
Cadmium	132/231	0.53	438	0.6–4.3	0.12	0.88	4	20	20	85/231	51/231	51/231
Chromium	24/26	1.03	33.2	5.8	1.57	19.7	1	0.4	10	24/26	24/26	9/26
Cobalt	25/26	0.5	35	5.8	0.96	28.3	20	--	1,000	4/26	--	0/26
Copper	26/26	2.88	58.3	-	4.34	16.9	100	50	100	0/26	3/26	0/26
Fluoride	1/13	3.8	3.8	0.7	0.4 [d]	0.4 [d]	200	--	30	0/13	--	0/13
Iron	32/32	593	181,000	-	912	45,100	--	--	200	--	--	32/32
Lead	175/271	12.1	16,000	14–110.9	2.9	23.3	50	500	900	122/271	54/271	43/271
Manganese	26/26	28.6	3,400	-	33.5	6,620	500	--	100	16/26	--	23/26
Mercury	13/13	0.1	4.16	-	0.07	0.15	0.3	0.1	30	6/13	12/13	0/13
Molybdenum	14/26	0.59	3.9	1.5–5.8	0.34	2.27	2	--	200	4/26	--	0/26
Nickel	25/26	1.58	37.5	11.5	4.33	36.8	30	200	90	6/26	0/26	0/26
Selenium	13/26	0.3	3.3	15–58	0.4	1	1	70	100	6/26	0/26	0/26
Silver	17/26	0.04	14.7	1.5–5.8	0.07	0.35	2	--	50	6/26	--	0/26
Strontium	18/18	4.8	150	-	7.3	39.6	--	--	--	--	--	--
Thallium	13/13	0.014	1.58	-	0.024	0.116	1	--	--	3/13	--	--
Tin	6/18	7.7	14	4.2–29	5	17.4	50	--	2,000	0/18	--	0/18
Vanadium	25/26	0.7	46.5	5.8	1.3	24.7	2	--	20	22/26	--	7/26
Zinc	271/271	15	82,700	-	47.8	111	50	200	100	244/271	135/271	181/271

Note: Field duplicates were screened separately.

- \- detected in all samples
-- \- no benchmark
ORNL \- Oak Ridge National Laboratory

[a] Expressed as the ratio of the detected exceedances over the total analyses.

[b] Efroymson et al. (1997a).

[c] Efroymson et al. (1997b).

[d] Undetected; value listed is one-half of the detection limit.

Table 3-20. Ecological screening results for stream sediment

					Reference Concentration		Ecological Screening Benchmark			Detection Frequency Above Benchmark[a]		
Analyte	Detection Frequency	Minimum Detected Concentration (mg/kg)	Maximum Detected Concentration (mg/kg)	Detection Limits (mg/kg)	Minimum Detected Concentration (mg/kg)	Maximum Detected Concentration (mg/kg)	Threshold Effect Concentration[b] (mg/kg)	Probable Effect Concentration[b] (mg/kg)	No Effect Concentration[c] (mg/kg)	Threshold Effect Concentration	Probable Effect Concentration	No Effect Concentration
Aluminum	15/15	4,080	17,100	-	3,620	12,100	--	--	73,000	--	--	0/15
Antimony	15/15	0.05	0.64	-	0.03	0.05	--	--	--	--	--	--
Arsenic	15/15	3.3	11.4	-	3.5	8.1	9.79	33	100	1/15	0/15	0/15
Barium	15/15	91.2	922	-	135	483	--	--	--	--	--	--
Cadmium	15/15	0.18	1.38	-	0.07	0.3	0.99	4.98	8	1/15	0/15	0/15
Chromium	15/15	7.35	23.3	-	7.22	19.9	43.4	111	95	0/15	0/15	0/15
Cobalt	15/15	7.9	17.6	-	7.3	11	--	--	--	--	--	--
Copper	15/15	9.66	28.2	-	5.99	18.5	31.6	149	580	0/15	0/15	0/15
Fluoride	0/15	0.95 [d]	1.2 [d]	1.9–2.3	1.2 [d]	1.2 [d]	--	--	--	--	--	--
Iron	15/15	22,800	45,700	-	21,300	27,300	--	--	290,000	--	--	0/15
Lead	15/15	8.24	142	-	5.05	9.17	35.8	128	130	2/15	2/15	1/15
Manganese	15/15	471	2140	-	268	859	--	--	4,500	--	--	0/15
Mercury	15/15	0.02	0.089	-	0.02	0.04	0.18	1.06	--	0/15	0/15	--
Molybdenum	15/15	0.34	2.32	-	0.28	0.52	--	--	--	--	--	--
Nickel	15/15	24.8	57.3	-	20.8	35	22.7	48.6	43	15/15	1/15	7/15
Selenium	15/15	0.4	2.5	-	0.4	0.7	--	--	--	--	--	--
Silver	15/15	0.05	0.42	-	0.03	0.12	--	--	--	--	--	--
Strontium	15/15	11	155	-	4.9	15	--	--	--	--	--	--
Thallium	15/15	0.031	0.322	-	0.023	0.07	--	--	--	--	--	--
Tin	4/15	4.3	7.6	1–5	2 [d]	2.4 [d]	--	--	--	--	--	--
Vanadium	15/15	8.83	28.1	-	10.7	24.8	--	--	--	--	--	--
Zinc	15/15	58.4	259	-	43.7	69.7	121	459	1,300	7/15	0/15	0/15

Note: Field duplicates were screened separately.

- - detected in all samples
-- - no benchmark

[a] Expressed as the ratio of the detected exceedances over the total analyses.

[b] MacDonald et al. (2000).

[c] Ingersoll et al. (1996).

[d] Undetected; value listed is one-half of the detection limit.

Table 3-21. Ecological screening results for stream surface water

Analyte	Detection Frequency	Minimum Detected Concentration (µg/L unfiltered)	Maximum Detected Concentration (µg/L unfiltered)	Detection Limits (µg/L)	Reference Concentration: Minimum Detected Concentration (µg/L unfiltered)	Reference Concentration: Maximum Detected Concentration (µg/L unfiltered)	Ecological Screening Benchmark: Freshwater Criteria Continuous Concentration[b] (µg/L total recoverable)	Ecological Screening Benchmark: Freshwater Criterion Maximum Concentration[b] (µg/L total recoverable)	Detection Frequency Above Benchmark[a]: Freshwater Criteria Continuous Concentration	Detection Frequency Above Benchmark[a]: Freshwater Criterion Maximum Concentration
Hardness	231/231	11,300	211,000	-	56,000	112,000	--	--	--	--
Aluminum	126/230	6.45	4060	5.04–62.1	17.3	2,770	87	750	29/230	3/230
Antimony	6/14	0.14	0.63	0.126	0.08	0.08	--	--	--	--
Arsenic	0/14	0.482 [c]	0.482 [c]	0.964	2.2	2.2	150	340	0/14[d]	0/14[d]
Barium	14/14	12.2	266	-	86.1	222	--	--	--	--
Cadmium	24/229	0.03	0.8	0.04–0.5	0.01	0.07	0.27 [e]	2.1 [e]	3/229	0/229
Chromium	0/18	0.396 [c]	0.4 [c]	0.791–0.8	0.17	3.71	11	16	0/18[d]	0/18[d]
Cobalt	11/14	0.03	0.33	0.02–0.03	0.12	2.72	--	--	--	--
Copper	16/18	0.3	1.23	0.21	0.6	5.4	9.4 [e]	14 [e]	0/18	0/18
Fluoride	27/31	40	120	100	30	40	--	--	--	--
Iron	186/230	6	10300	5.13–50	64.2	6,710	1,000	--	12/230	--
Lead	84/230	0.018	7.34	0.04–0.802	0.02	1.91	3.2 [e]	82 [e]	22/230	0/230
Manganese	17/18	0.56	36	0.95	4.87	128	--	--	--	--
Mercury	0/14	0.0179 [c]	0.0179 [c]	0.0358	0.05 [c]	0.05 [c]	0.91	1.6	0/14[d]	0/14[d]
Molybdenum	9/14	0.37	2.27	0.355–0.48	0.05	0.17	--	--	--	--
Nickel	14/14	0.26	6.71	-	1.06	10.5	52 [e]	470 [e]	0/14	0/14
Selenium	15/29	0.0666	1.24	0.0402	0.2 [c]	0.2 [c]	5.0	--	0/29	--
Silver	0/14	0.023 [c]	0.023 [c]	0.046	0.03	0.03	--	3.8 [e]	--	0/14[d]
Strontium	14/14	19.4	172	-	32.5	81.1	--	--	--	--
Thallium	4/29	0.04	0.55	0.031–0.29	0.014	0.014	--	--	--	--
Tin	5/14	1.3	5.33	1.18	10 [c]	10 [c]	--	--	--	--
Vanadium	4/14	0.67	0.93	0.669	0.16	5.57	--	--	--	--
Zinc	107/230	1	60.1	1–10	0.31	9.84	120 [e]	120 [e]	1/230	1/230

Note: Field duplicates were screened separately.

- - detected in all samples
-- - no benchmark

[a] Expressed as the ratio of the detected exceedances over the total analyses.

[b] U.S. EPA (2002c).

[c] Undetected; value listed is one-half of the detection limit.

[d] Undetected in all samples. Undetected values expressed as one-half of the detection limit are below the screening benchmark.

[e] Benchmark is hardness-dependent. Value listed corresponds to a hardness of 100 mg/L $CaCO_3$. Benchmark applied in the screening was adjusted for hardness.

Table 3-22. Ecological screening results for tundra pond sediment

Analyte	Detection Frequency	Minimum Detected Concentration (mg/kg)	Maximum Detected Concentration (mg/kg)	Detection Limits (mg/kg)	Reference Concentration: Minimum Detected Concentration (mg/kg)	Reference Concentration: Maximum Detected Concentration (mg/kg)	Ecological Screening Benchmark: Threshold Effect Concentration[b] (mg/kg)	Ecological Screening Benchmark: Probable Effect Concentration[b] (mg/kg)	Ecological Screening Benchmark: No Effect Concentration[c] (mg/kg)	Detection Frequency Above Benchmark[a]: Threshold Effect Concentration	Detection Frequency Above Benchmark[a]: Probable Effect Concentration	Detection Frequency Above Benchmark[a]: No Effect Concentration
Aluminum	5/5	1,920	5,270	-	3,730	17,100	--	--	73,000	--	--	0/5
Antimony	5/5	0.19	10.9	-	0.03	0.11	--	--	--	--	--	--
Arsenic	5/5	2.6	8.7	-	2.6	13	9.79	33	100	0/5	0/5	0/5
Barium	5/5	281	626	-	121	772	--	--	--	--	--	--
Cadmium	5/5	0.93	119	-	0.27	0.66	0.99	4.98	8	3/5	2/5	2/5
Chromium	5/5	8.97	15.3	-	9.57	28	43.4	111	95	0/5	0/5	0/5
Cobalt	5/5	2.66	25.9	-	1.83	21.9	--	--	--	--	--	--
Copper	5/5	6.51	53.4	-	7.99	20.7	31.6	149	580	2/5	0/5	0/5
Fluoride	3/5	2.6	4.4	2.3	1.2 [d]	1.2 [d]	--	--	--	--	--	--
Iron	5/5	16,000	51,900	-	17,900	43,700	--	--	290,000	--	--	0/5
Lead	5/5	8.96	2,180	-	7.48	20.3	35.8	128	130	3/5	2/5	2/5
Manganese	5/5	60.2	745	-	15.9	1,870	--	--	4,500	--	--	0/5
Mercury	5/5	0.06	1.31	-	0.03	0.07	0.18	1.06	--	2/5	1/5	--
Molybdenum	5/5	1.05	2.84	-	0.38	1.35	--	--	--	--	--	--
Nickel	5/5	17.6	44.2	-	12	70.3	22.7	48.6	43	4/5	0/5	1/5
Selenium	4/5	1.2	3.5	1.5	0.5	3.1	--	--	--	--	--	--
Silver	3/5	0.09	3.76	0.1–0.15	0.06	0.18	--	--	--	--	--	--
Strontium	5/5	17.1	111	-	4.2	25.4	--	--	--	--	--	--
Thallium	4/5	0.021	1.92	0.046	0.056	0.174	--	--	--	--	--	--
Tin	3/5	18.2	41.2	15.4–21.3	2.1 [d]	6.3 [d]	--	--	--	--	--	--
Vanadium	5/5	10.2	28.3	-	14.9	94.5	--	--	--	--	--	--
Zinc	5/5	143	27,000	-	23.4	138	121	459	1,300	5/5	2/5	2/5

Note: Field duplicates were screened separately.

- - detected in all samples
-- - no benchmark

[a] Expressed as the ratio of the detected exceedances over the total analyses.

[b] MacDonald et al. (2000).

[c] Ingersoll et al. (1996).

[d] Undetected; value listed is one-half of the detection limit.

Table 3-23. Ecological screening results for tundra pond surface water

Analyte	Detection Frequency	Minimum Detected Concentration (µg/L unfiltered)	Maximum Detected Concentration (µg/L unfiltered)	Detection Limits (µg/L)	Reference Concentration: Minimum Detected Concentration (µg/L unfiltered)	Reference Concentration: Maximum Detected Concentration (µg/L unfiltered)	Ecological Screening Benchmark: Freshwater Criteria Continuous Concentration[b] (µg/L total recoverable)	Ecological Screening Benchmark: Freshwater Criterion Maximum Concentration[b] (µg/L total recoverable)	Detection Frequency Above Benchmark[a]: Freshwater Criteria Continuous Concentration	Detection Frequency Above Benchmark[a]: Freshwater Criterion Maximum Concentration
Hardness	5/5	10,300	382,000	-	12,300	34,400	--	--	--	--
Aluminum	5/5	11.4	180	-	14.5	170	87	750	3/5	0/5
Antimony	3/5	0.03	0.2	0.04	0.02	0.1	--	--	--	--
Arsenic	5/5	0.4	1.7	-	0.5	0.9	150	340	0/5	0/5
Barium	5/5	39.4	74.5	-	48.4	133	--	--	--	--
Cadmium	5/5	0.02	0.27	-	0.05	0.06	0.27 [c]	2.1 [c]	1/5	0/5
Chromium	5/5	0.44	6.31	-	0.18	1.98	11	16	0/5	0/5
Cobalt	5/5	0.13	1.56	-	0.19	0.7	--	--	--	--
Copper	5/5	0.4	2.8	-	0.7	2.5	9.4 [c]	14 [c]	2/5	0/5
Fluoride	5/5	20	60	-	20	50	--	--	--	--
Iron	5/5	685	1,220	-	361	1,500	1,000	--	4/5	--
Lead	5/5	0.44	1.63	-	0.06	0.56	3.2 [c]	82 [c]	4/5	0/5
Manganese	5/5	2.87	132	-	4.22	71.2	--	--	--	--
Mercury	0/5	0.05 [d]	0.05 [d]	0.1	0.05 [d]	0.05 [d]	0.91	1.6	0/5[e]	0/5[e]
Molybdenum	4/5	0.05	0.09	0.03	0.05	0.22	--	--	--	--
Nickel	5/5	2.96	5.41	-	2.11	6.41	52 [c]	470 [c]	0/5	0/5
Selenium	0/5	0.2 [d]	0.2 [d]	0.3	0.3	0.5	5.0	--	0/5[e]	--
Silver	0/5	0.005 [d]	0.01 [d]	0.01–0.02	0.04	0.04	--	3.8 [c]	--	0/5[e]
Strontium	5/5	10.4	422	-	10.6	27.5	--	--	--	--
Thallium	1/5	0.01	0.01	0.005–0.01	0.04	0.04	--	--	--	--
Tin	1/5	30	30	20	10 [d]	10 [d]	--	--	--	--
Vanadium	5/5	0.24	0.65	-	0.17	2.41	--	--	--	--
Zinc	5/5	6.08	99	-	0.59	5.01	120 [c]	120 [c]	1/5	1/5

Note: Field duplicates were screened separately.

- - detected in all samples
-- - no benchmark

[a] Expressed as the ratio of the detected exceedances over the total analyses.

[b] U.S. EPA (2002c).

[c] Benchmark is hardness-dependent. Value listed corresponds to a hardness of 100 mg/L $CaCO_3$. Benchmark applied in the screening was adjusted for hardness.

[d] Undetected; value listed is one-half of the detection limit.

[e] Undetected in all samples. Undetected values expressed as one-half of the detection limit are below the screening benchmark.

Table 3-24. Ecological screening results for lagoon sediment

Analyte	Detection Frequency	Minimum Detected Concentration (mg/kg)	Maximum Detected Concentration (mg/kg)	Detection Limits (mg/kg)	Reference Concentration: Minimum Detected Concentration (mg/kg)	Reference Concentration: Maximum Detected Concentration (mg/kg)	Ecological Screening Benchmark: Effects Range Low[b] (mg/kg)	Ecological Screening Benchmark: Effects Range Median[b] (mg/kg)	Ecological Screening Benchmark: Marine Sediment Quality Standards[c] (mg/kg)	Detection Frequency Above Benchmark[a]: Effects Range Low	Detection Frequency Above Benchmark[a]: Effects Range Median	Detection Frequency Above Benchmark[a]: Marine Sediment Quality Standards
Aluminum	9/9	2,450	15,300	-	7,440	14,800	--	--	--	--	--	--
Antimony	9/9	0.07	0.27	-	0.1	0.12	--	--	--	--	--	--
Arsenic	9/9	4.9	17.9	-	2.6	4.9	8.2	70	57	2/9	0/9	0/9
Barium	9/9	54.1	350	-	164	271	--	--	--	--	--	--
Cadmium	14/29	0.03	8.1	0.8–5.1	0.18	0.49	1.2	9.6	5.1	6/29	0/29	1/29
Chromium	9/9	4.08	27.2	-	12.5	24.9	81	370	260	0/9	0/9	0/9
Cobalt	9/9	3.85	11.8	-	4.97	9.68	--	--	--	--	--	--
Copper	9/9	3	28.2	-	9.87	18.7	34	270	390	0/9	0/9	0/9
Fluoride	2/9	5.1	8.6	1.9–2.3	1.2 [d]	1.2 [d]	--	--	--	--	--	--
Iron	9/9	10,100	75,000	-	14,000	22,200	--	--	--	--	--	--
Lead	28/29	4.66	302	21	3.2	23	46.7	218	450	8/29	1/29	0/29
Manganese	9/9	97.9	274	-	75.5	129	--	--	--	--	--	--
Mercury	8/9	0.01	0.096	0.008	0.03	0.06	0.15	0.71	0.41	0/9	0/9	0/9
Molybdenum	9/9	0.41	3.39	-	0.46	0.98	--	--	--	--	--	--
Nickel	9/9	12	39	-	18.7	37	20.9	51.6	--	5/9	0/9	--
Selenium	8/9	0.3	2.2	0.2	0.6	1.4	--	--	--	--	--	--
Silver	9/9	0.03	0.27	-	0.08	0.11	1.0	3.7	6.1	0/9	0/9	0/9
Strontium	9/9	10.4	108	-	20.9	40	--	--	--	--	--	--
Thallium	9/9	0.018	0.184	-	0.038	0.103	--	--	--	--	--	--
Tin	1/9	6.7	6.7	1.3–5.3	4.2	5.1	--	--	--	--	--	--
Vanadium	9/9	8.5	35.1	-	16.8	31.5	--	--	--	--	--	--
Zinc	29/29	36	1,590	-	16	370.6	150	410	410	14/29	5/29	5/29

Note: Field duplicates were screened separately.

- - detected in all samples
-- - no benchmark

[a] Expressed as the ratio of the detected exceedances over the total analyses.

[b] Long et al. (1995).

[c] WAC 173-204-320.

[d] Undetected; value listed is one-half of the detection limit.

Table 3-25. Ecological screening results for lagoon surface water

Analyte	Detection Frequency	Minimum Detected Concentration (µg/L unfiltered)	Maximum Detected Concentration (µg/L unfiltered)	Detection Limits (µg/L)	Reference Concentration: Minimum Detected Concentration (µg/L unfiltered)	Reference Concentration: Maximum Detected Concentration (µg/L unfiltered)	Ecological Screening Benchmark: Saltwater Criteria Continuous Concentration[b] (µg/L total recoverable)	Ecological Screening Benchmark: Saltwater Criterion Maximum Concentration[b] (µg/L total recoverable)	Detection Frequency Above Benchmark[a]: Saltwater Criteria Continuous Concentration	Detection Frequency Above Benchmark[a]: Saltwater Criterion Maximum Concentration
Aluminum	9/9	19.7	247	-	53.5	434	--	--	--	--
Antimony	9/9	0.19	0.63	-	0.11	0.13	--	--	--	--
Arsenic	9/9	4.4	126	-	52.9	98.8	36	69	4/9	3/9
Barium	9/9	111	413	-	144	168	--	--	--	--
Cadmium	12/15	0.05	0.30	0.1	0.18	0.26	8.9	40	0/15	0/15
Chromium	9/9	1.63	4.49	-	5.96	8.22	50	1,100	0/9	0/9
Cobalt	9/9	0.44	1.38	-	3.7	5.35	--	--	--	--
Copper	9/9	0.5	1.4	-	0.4	1.4	3.7	5.8	0/9	0/9
Fluoride	9/9	50	200	-	20	20	--	--	--	--
Iron	9/9	200	723	-	290	693	--	--	--	--
Lead	15/15	0.4	2.3	-	0.1	0.85	8.5	220	0/15	0/15
Manganese	9/9	13.9	277	-	492	801	--	--	--	--
Mercury	0/9	0.05 [c]	0.05 [c]	0.1	0.05 [c]	0.05 [c]	1.1	2.1	0/9[d]	0/9[d]
Molybdenum	9/9	0.3	2.41	-	0.07	0.09	--	--	--	--
Nickel	9/9	3.42	10.6	-	9.19	15.2	8.3	75	2/9	0/9
Selenium	6/9	0.3	0.6	0.3	0.2 [c]	0.2 [c]	71	290	0/9	0/9
Silver	8/9	0.01	0.25	0.1	0.02	0.03	--	2.2	--	0/9
Strontium	9/9	503	1,850	-	991	1,470	--	--	--	--
Thallium	5/9	0.007	0.07	0.025–0.05	0.006	0.009	--	--	--	--
Tin	1/9	23.7	23.7	20	10 [c]	10 [c]	--	--	--	--
Vanadium	6/9	0.22	0.85	0.35–0.7	0.4 [c]	0.4 [c]	--	--	--	--
Zinc	15/15	3.09	110	-	17	30.1	86	95	1/15	1/15

Note: Field duplicates were screened separately.

- - detected in all samples
-- - no benchmark

[a] Expressed as the ratio of the detected exceedances over the total analyses.

[b] U.S. EPA (2002c).

[c] Undetected; value listed is one-half of the detection limit.

[d] Undetected in all samples. Undetected values expressed as one-half of the detection limit are below the screening benchmark.

Table 3-26. Ecological screening results for marine sediment

Analyte	Detection Frequency	Minimum Detected Concentration (mg/kg)	Maximum Detected Concentration (mg/kg)	Detection Limits (mg/kg)	Reference Concentration: Minimum Detected Concentration (mg/kg)	Reference Concentration: Maximum Detected Concentration (mg/kg)	Ecological Screening Benchmark: Effects Range Low[b] (mg/kg)	Ecological Screening Benchmark: Effects Range Median[b] (mg/kg)	Ecological Screening Benchmark: Marine Sediment Quality Standards[c] (mg/kg)	Detection Frequency Above Benchmark[a]: Effects Range Low	Detection Frequency Above Benchmark[a]: Effects Range Median	Detection Frequency Above Benchmark[a]: Marine Sediment Quality Standards
Aluminum	20/20	1,990	6,070	-	1,970	8,000	--	--	--	--	--	--
Antimony	16/43	0.06	4.59	0.08–14	0.07	0.13	--	--	--	--	--	--
Arsenic	70/71	3.1	14.5	27	5.6	13	8.2	70	57	13/71	0/71	0/71
Barium	71/71	79.5	639	-	22	431	--	--	--	--	--	--
Cadmium	107/136	0.04	52.9	0.03–1.5	0.02	0.23	1.2	9.6	5.1	24/136	1/136	1/136
Chromium	71/71	2.4	33.5	-	1.4	18	81	370	260	0/71	0/71	0/71
Cobalt	20/20	3.15	8.89	-	4.2	8.71	--	--	--	--	--	--
Copper	71/71	3.7	34.8	-	3	10.2	34	270	390	1/71	0/71	0/71
Fluoride	18/18	0.4	1.5	-	0.4	2	--	--	--	--	--	--
Iron	20/20	9,960	19,300	-	8,150	22,700	--	--	--	--	--	--
Lead	136/136	1.59	5,620	-	2.7	11.2	46.7	218	450	5/136	1/136	1/136
Manganese	20/20	161	363	-	187	389	--	--	--	--	--	--
Mercury	35/69	0.01	0.58	0.01–0.2	0.01	0.02	0.15	0.71	0.41	1/69	0/69	1/69
Molybdenum	19/20	0.37	1.04	2.7	0.44	0.83	--	--	--	--	--	--
Nickel	20/20	11.3	33.6	-	9.8	34.8	20.9	51.6	--	15/20	0/20	--
Selenium	26/71	0.3	6	1–27	0.3	1.2	--	--	--	--	--	--
Silver	42/71	0.03	2.11	0.05–2.9	0.02	0.49	1.0	3.7	6.1	1/71	0/71	0/71
Strontium	19/19	24.4	33.8	-	13	29	--	--	--	--	--	--
Thallium	19/19	0.023	1.13	-	0.025	0.052	--	--	--	--	--	--
Tin	5/19	5.6	8.9	1.9–5	4.8	9.5	--	--	--	--	--	--
Vanadium	43/43	9.11	46	-	13	33.9	--	--	--	--	--	--
Zinc	136/136	5.5	2,550	-	25	56.8	150	410	410	7/136	3/136	3/136

Note: Field duplicates were screened separately.

- - detected in all samples
-- - no benchmark

[a] Expressed as the ratio of the detected exceedances over the total analyses.

[b] Long et al. (1995).

[c] WAC 173-204-320.

Table 3-27. Ecological screening results for marine surface water

Analyte	Detection Frequency	Minimum Detected Concentration (µg/L unfiltered)	Maximum Detected Concentration (µg/L unfiltered)	Detection Limits (µg/L)	Reference Concentration: Minimum Detected Concentration (µg/L unfiltered)	Reference Concentration: Maximum Detected Concentration (µg/L unfiltered)	Ecological Screening Benchmark: Saltwater Criteria Continuous Concentration[b] (µg/L total recoverable)	Ecological Screening Benchmark: Saltwater Criterion Maximum Concentration[b] (µg/L total recoverable)	Detection Frequency Above Benchmark[a]: Saltwater Criteria Continuous Concentration	Detection Frequency Above Benchmark[a]: Saltwater Criterion Maximum Concentration
Aluminum	9/11	44	215	50	63.9	336	--	--	--	--
Antimony	6/11	0.47	2.79	0.4–0.8	0.44	1.67	--	--	--	--
Arsenic	8/11	1.5	6.3	5	1.1	7.5	36	69	0/11	0/11
Barium	11/11	12.1	39.4	-	9.91	38.1	--	--	--	--
Cadmium	11/11	1.58	4.62	-	2.27	4.69	8.9	40	0/11	0/11
Chromium	0/11	1 [c]	2 [c]	2-3	1 [c]	2 [c]	50	1,100	0/11[d]	0/11[d]
Cobalt	11/11	3.85	4.6	-	4.03	4.48	--	--	--	--
Copper	5/11	1	5.9	8	1	2.6	3.7	5.8	1/11	1/11
Fluoride	11/11	500	900	-	600	800	--	--	--	--
Iron	11/11	52.3	375	-	33.6	643	--	--	--	--
Lead	11/11	0.7	1.47	-	0.76	1.25	8.5	220	0/11	0/11
Manganese	11/11	13.1	31.9	-	10.1	25.5	--	--	--	--
Mercury	0/11	0.05 [c]	0.05 [c]	0.1	0.05 [c]	0.05 [c]	1.1	2.1	0/11[d]	0/11[d]
Molybdenum	11/11	8.1	11.1	-	8.26	10.6	--	--	--	--
Nickel	0/11	2 [c]	10 [c]	3–20	2 [c]	10 [c]	8.3	75	0/11[e]	0/11[e]
Selenium	9/11	0.3	1	0.3	0.3	0.5	71	290	0/11	0/11
Silver	5/11	0.51	0.95	0.2–0.4	0.27	0.27	--	2.2	--	0/11
Strontium	11/11	4,310	5,650	-	4,530	5,290	--	--	--	--
Thallium	1/11	0.09	0.09	0.1–0.2	0.111	0.133	--	--	--	--
Tin	1/11	23.3	23.3	6-20	26.4	26.4	--	--	--	--
Vanadium	2/11	4.44	5.27	2.8–4	3.77	8.44	--	--	--	--
Zinc	0/11	1 [c]	1 [c]	2	1 [c]	1 [c]	86	95	0/11[d]	0/11[d]

Note: Field duplicates were screened separately.

- - detected in all samples
-- - no benchmark

[a] Expressed as the ratio of the detected exceedances over the total analyses.

[b] U.S. EPA (2002c).

[c] Undetected; value listed is one-half of the detection limit.

[d] Undetected in all samples. Undetected values expressed as one-half of the detection limit are below the screening benchmark.

[e] Undetected in all samples. Five undetected values expressed as one-half of the detection limit exceed the benchmark.

Table 3-28. Toxicity reference values for risk evaluation for wildlife receptors

	TRVs (mg/kg-day)					
	Avian			Mammalian		
CoPC	NOAEL	LOAEL	Citation	NOAEL	LOAEL	Citation
Aluminum	120	NA	Carriere et al. (1986)	1.9	19	Ondreicka et al. (1966)
Antimony	NA	NA	NA	0.66	NA	Schroeder et al. (1970)
Arsenic[a]	10	40	Stanley et al. (1994)	0.13	1.3	Schroeder and Mitchener (1971)
Barium	21	42	Johnson et al. (1960)	5.1	--	Perry et al. (1983)
	--	--	--	--	20	Borzelleca et al. (1988)
Cadmium	1.5	20	White and Finley (1978)	1.0	10	Sutou et al. (1980)
Chromimum	0.86	4.3	Haseltine et al. (1985) as cited in Sample et al. (1996)	3.3	--	Mackenzie et al. (1958)
	--	--	--	--	69	Gross and Heller (1946)
Cobalt	NA	NA	NA	0.5	2.0	Nation et al. (1983)
Copper	47	62	Mehring et al. (1960)	12	15	Aulerich et al. (1982)
Fluoride	7.8	32	Pattee et al. (1988)	31	53	Aulerich et al. (1987)
Iron	NA	NA	NA	NA	NA	NA
Lead	3.9	NA	Pattee (1984)	11	90	Azar et al. (1973)
Manganese	980	NA	Laskey and Edens (1985)	88	280	Laskey et al. (1982)
Mercury[b]	0.032	0.064	Heinz (1974, 1976a,b, 1979)	0.032	0.16	Verschuuren et al. (1976)
Molybdenum	3.5	35	Lepore and Miller (1965)	0.26	2.6	Schroeder and Mitchener (1971)
Nickel	77	110	Cain and Pafford (1981)	40	80	Ambrose et al. (1976)
Selenium	0.40	0.80	Heinz et al. (1989)	0.20	0.33	Rosenfeld and Beath (1954)
Silver	NA	NA	NA	NA	NA	NA
Strontium	NA	NA	NA	263	NA	Skoryna (1981)
Thallium	0.24	24	Hudson et al. (1984)	0.074	0.74	Formigli et al. (1986)
Tin[c]	6.8	17	Schlatterer et al. (1993)	23	35	Davis et al. (1987)
Vanadium	11	NA	White and Dieter (1978)	0.21	2.1	Domingo et al. (1986)
Zinc	130	NA	Stahl et al. (1990)	160	320	Schlicker and Cox (1968)

Note: CoPC - chemical of potential concern
LOAEL - lowest-observed-adverse-effect level
NA - not available; no suitable TRV was derived
NOAEL - no-observed-adverse-effect level
TRV - toxicity reference value

[a] Avian TRVs were based on exposure to arsenic as arsenate; mammalian TRVs were based on exposure to arsenic as arsenite.

[b] Mercury TRVs were based on exposure to methylmercury.

[c] Tin TRVs were based on exposure to tributyltin.

Table 3-29. Ecological exposure assumptions for use in screening food-web models

Representative Receptor	Body Weight (kg)	Food Ingestion Rate kg/day (dry wt)	Soil/Sediment Ingestion Rate kg/day (dry wt)	Area Use Factor
Tundra vole	0.029 [a]	0.0060 [b]	0.00014 [c]	1
Common snipe	0.081 [d]	0.012 [e]	0.0012 [f]	1
Red-throated loon	1.15 [g]	0.079 [h]	0.0016 [i]	1
River otter	6.8 [j]	0.19 [k]	0.018 [l]	1
Black-bellied plover	0.19 [m]	0.026 [n]	0.0075 [o]	1

[a] Minimum female body weight from Bee and Hall (1956).

[b] Based on Nagy et al. (1999) allometric equation for Rodentia.

[c] Based on 2.4 percent soil in meadow vole diet from Beyer et al. (1994).

[d] Minimum female body weight from Tuck (1972).

[e] Based on Nagy et al. (1999) allometric equation for insectivores.

[f] Based on 10.4 percent soil in American woodcock diet from Beyer et al. (1994).

[g] Minimum body weight from Dunning (1993).

[h] Based on Nagy et al. (1999) allometric equation for all birds.

[i] Based on minimum soil ingestion rate from Beyer et al. (1994).

[j] Minimum body weight from DFG (2002).

[k] Based on Nagy et al. (1999) allometric equation for Carnivora.

[l] Based on 9.4 percent soil in raccoon diet from Beyer et al. (1994).

[m] Minimum female body weight for Alaska from Paulson (1995).

[n] Based on Nagy et al. (1999) allometric equation for Charadriiformes.

[o] Based on 29 percent sediment in black-bellied plover diet from Hui and Beyer (1998).

Table 3-30. Screening-level food-web results for tundra vole

Analyte	Maximum Concentration Tundra Soil (mg/kg dw)	Maximum Concentration Moss (mg/kg dw)	Daily Exposure Tundra Soil (mg/day)	Daily Exposure Moss (mg/day)	Total Daily Intake (mg/day)	Body Weight Normalized Exposure (mg/kg-day)	TRV (mg/kg-day)	Hazard Quotient
Site								
Metals								
Aluminum	18,900	47,900	2.73	289	291	10,000	1.9	5,300
Antimony	25.8	4.58	0.00373	0.0276	0.0313	1.08	0.66	1.6
Arsenic	150	15.7	0.0217	0.0946	0.116	4.01	0.13	31
Barium	5,810	8,800	0.840	53.0	53.9	1,857	5.1	360
Cadmium	438	48.4	0.0633	0.292	0.355	12.2	1.0	12
Chromium	33.2	32.9	0.00480	0.198	0.203	7.00	3.3	2.1
Cobalt	35	9.35	0.00506	0.0563	0.0614	2.12	0.50	4.2
Copper	58.3	40.5	0.00843	0.244	0.252	8.71	12	0.73
Fluoride	3.8	NA	0.000550	NA	0.000550	0.0189	31	0.000611
Lead	16,000	1,720	2.31	10.4	12.7	437	11	40
Manganese	3,400	842	0.492	5.07	5.57	192	88	2.2
Mercury	4.16	1.04	0.000602	0.00627	0.00687	0.237	0.032	7.4
Molybdenum	3.9	2.4	0.000564	0.0145	0.0150	0.518	0.26	2.0
Nickel	37.5	31.6	0.00542	0.190	0.196	6.75	40	0.17
Selenium	3	1.5	0.000477	0.00904	0.00952	0.328	0.20	1.6
Strontium	150	107	0.0217	0.645	0.666	23.0	260	0.088
Thallium	1.58	1.84	0.000228	0.0111	0.0113	0.390	0.074	5.3
Tin	14	3.9	0.00202	0.0235	0.0255	0.880	23	0.038
Vanadium	46.5	14.7	0.00672	0.0886	0.0953	3.29	0.21	16
Zinc	82,700	8,120	12.0	48.9	60.9	2,100	160	13
Reference								
Metals								
Aluminum	11,300	713	1.63	4.30	5.93	204	1.9	110
Antimony	0.28	0.15	0.0000405	0.000904	0.000944	0.0326	0.66	0.049
Arsenic	6.8	0.3	0.000983	0.00181	0.00279	0.0962	0.13	0.74
Barium	624	119	0.0902	0.717	0.807	27.8	5.1	5.5
Cadmium	0.88	0.38	0.000127	0.00229	0.00242	0.0833	1.0	0.083
Chromium	19.7	2.96	0.00285	0.0178	0.0207	0.713	3.3	0.22
Cobalt	28.3	2.03	0.00409	0.0122	0.0163	0.563	0.50	1.1
Copper	16.9	4.35	0.00244	0.0262	0.0287	0.988	12	0.082
Fluoride	0.4	NA	0.0000578	NA	0.0000578	0.00199	31	0.000064
Lead	23.3	9.64	0.00337	0.0581	0.0615	2.12	11	0.19
Manganese	6,620	712	0.957	4.29	5.25	181	88	2.1
Mercury	0.15	0.067	0.0000217	0.000404	0.000425	0.0147	0.032	0.46
Molybdenum	2.27	0.3	0.000328	0.00181	0.00214	0.0737	0.26	0.28
Nickel	36.8	6.34	0.00532	0.0382	0.0435	1.50	40	0.038
Selenium	1	0.1	0.000145	0.000603	0.000747	0.0258	0.20	0.13
Strontium	39.6	11	0.00573	0.0663	0.0720	2.48	260	0.0096
Thallium	0.116	0.04	0.0000168	0.000241	0.000258	0.00889	0.074	0.12
Tin	17.4	1.1	0.00252	0.00663	0.00914	0.315	23	0.014
Vanadium	24.7	1.73	0.00357	0.0104	0.0140	0.483	0.21	2.3
Zinc	111	64	0.0161	0.386	0.402	13.9	160	0.087

Note: Hazard quotients greater than 1.0 are boxed.

NA - not available

TRV - toxicity reference value

Table 3-31. Screening-level food-web results for river otter

Analyte	Maximum Concentration Sediment (mg/kg dw)	Maximum Concentration Fish (mg/kg dw)	Daily Exposure Sediment (mg/day)	Daily Exposure Fish (mg/day)	Total Daily Intake	Body Weight Normalized Exposure	TRV (mg/kg-day)	Hazard Quotient
Aufeis Creek								
Metals								
Cadmium	0.31	0.17	0.00558	0.0325	0.0382	0.00562	1.0	0.0056
Lead	14.9	6	0.268	1.15	1.42	0.208	11	0.019
Selenium	2.5	7	0.0450	1.34	1.38	0.203	0.20	1.0
Zinc	136	121	2.45	23.1	25.7	3.77	160	0.024
Omikviorok River								
Metals								
Cadmium	0.59	0.14	0.0106	0.0268	0.0375	0.00551	1.0	0.0055
Lead	19	3.03	0.342	0.579	0.921	0.136	11	0.012
Selenium	0.7	3.4	0.0126	0.650	0.662	0.0974	0.20	0.49
Zinc	123	155	2.21	29.6	31.9	4.69	160	0.029
Anxiety Ridge Creek								
Metals								
Cadmium	1.38	0.39	0.0248	0.0745	0.0994	0.0146	1.0	0.015
Lead	142	2.86	2.56	0.547	3.10	0.456	11	0.041
Selenium	2.4	5.87	0.0432	1.12	1.17	0.171	0.20	0.86
Zinc	259	155 [a]	4.66	29.6	34.3	5.04	160	0.032

Note: Hazard quotients greater than 1.0 are boxed.

TRV - toxicity reference value

[a] No zinc tissue data available for Anxiety Ridge Creek; maximum concentration from Omikviorok River used in calculation.

Table 3-32. Screening-level food-web results for red-throated loon

Analyte	Maximum Concentration Sediment (mg/kg dw)	Maximum Concentration Fish (mg/kg dw)	Daily Exposure Sediment (mg/day)	Daily Exposure Fish (mg/day)	Total Daily Intake	Body Weight Normalized Exposure	TRV (mg/kg-day)	Hazard Quotient
Aufeis Creek								
Metals								
Cadmium	0.31	0.17	0.000488	0.0134	0.0139	0.0121	1.5	0.0080
Lead	14.9	6	0.0235	0.472	0.496	0.431	3.9	0.11
Selenium	2.5	7	0.00394	0.551	0.555	0.483	0.40	1.2
Zinc	136	121	0.214	9.52	9.74	8.47	130	0.065
Omikviorok River								
Metals								
Cadmium	0.59	0.14	0.000929	0.0110	0.0120	0.0104	1.5	0.0069
Lead	19	3.03	0.0299	0.238	0.269	0.234	3.9	0.060
Selenium	0.7	3.4	0.00110	0.268	0.269	0.234	0.40	0.58
Zinc	123	155	0.194	12.2	12.4	10.8	130	0.083
Anxiety Ridge Creek								
Metals								
Cadmium	1.38	0.39	0.00217	0.0307	0.0329	0.0286	1.5	0.019
Lead	142	2.86	0.224	0.225	0.449	0.390	3.9	0.10
Selenium	2.4	5.87	0.00378	0.462	0.466	0.405	0.40	1.0
Zinc	259	155 [a]	0.408	12.2	12.6	11.0	130	0.084

Note: Hazard quotients greater than 1.0 are boxed.

TRV - toxicity reference value

[a] No zinc tissue data available for Anxiety Ridge Creek; maximum concentration from Omikviorok River used in calculation.

Table 3-33. Screening-level food-web results for common snipe foraging in freshwater rivers and creeks

Analyte	Maximum Concentration: Sediment (mg/kg dw)	Maximum Concentration: Invertebrates (mg/kg dw)	Daily Exposure: Sediment (mg/day)	Daily Exposure: Invertebrates (mg/day)	Total Daily Intake	Body Weight Normalized Exposure	TRV (mg/kg-day)	Hazard Quotient
New Heart Creek								
Metals								
Aluminum	17,100	17,100	20.5	205	226	2,790	120	23
Arsenic	7.3	7.3	0.00876	0.0876	0.0964	1.19	10	0.12
Barium	293	293	0.352	3.52	3.87	47.7	21	2.3
Cadmium	0.77	0.77	0.000924	0.00924	0.0102	0.125	1.5	0.084
Chromium	15.6	15.6	0.0187	0.187	0.206	2.54	0.86	3.0
Copper	14.2	14.2	0.0170	0.170	0.187	2.31	47	0.049
Fluoride	1.2	1.2	0.00144	0.0144	0.0158	0.196	7.8	0.025
Lead	23.8	23.8	0.0286	0.286	0.314	3.88	3.9	0.99
Manganese	939	939	1.13	11.3	12.4	153	980	0.16
Mercury	0.06	0.06	0.0000720	0.000720	0.000792	0.00978	0.032	0.31
Molybdenum	0.84	0.84	0.00101	0.0101	0.0111	0.137	3.5	0.039
Nickel	45.2	45.2	0.0542	0.542	0.597	7.37	77	0.096
Selenium	1.4	1.4	0.00168	0.0168	0.0185	0.228	0.4	0.57
Thallium	0.08	0.08	0.0000960	0.000960	0.00106	0.0130	0.24	0.055
Tin	7.6	7.6	0.00912	0.0912	0.100	1.24	6.8	0.18
Vanadium	13.8	13.8	0.0166	0.166	0.182	2.25	11	0.20
Zinc	206	206	0.247	2.47	2.72	33.6	130	0.26
Aufeis Creek								
Metals								
Aluminum	7,580	7,580	9.10	91.0	100	1,240	120	10
Arsenic	9.6	9.6	0.0115	0.115	0.127	1.56	10	0.16
Barium	172	172	0.206	2.06	2.27	28.0	21	1.3
Cadmium	0.31	0.31	0.000372	0.00372	0.00409	0.0505	1.5	0.034
Chromium	22.1	22.1	0.0265	0.265	0.292	3.60	0.86	4.2
Copper	28.2	28.2	0.0338	0.338	0.372	4.60	47	0.098
Fluoride	1.2	1.2	0.00144	0.0144	0.0158	0.196	7.8	0.025
Lead	14.9	14.9	0.0179	0.179	0.197	2.43	3.9	0.62
Manganese	1,200	1,200	1.44	14.4	15.8	196	980	0.20
Mercury	0.06	0.06	0.0000720	0.000720	0.000792	0.00978	0.032	0.31
Molybdenum	1.01	1.01	0.00121	0.0121	0.0133	0.165	3.5	0.047
Nickel	46.5	46.5	0.0558	0.558	0.614	7.58	77	0.098
Selenium	2.5	2.5	0.00300	0.0300	0.0330	0.407	0.40	1.0
Thallium	0.115	0.115	0.000138	0.00138	0.00152	0.0187	0.24	0.079
Tin	6.4	6.4	0.00768	0.0768	0.0845	1.04	6.8	0.15
Vanadium	18.2	18.2	0.0218	0.218	0.240	2.97	11	0.27
Zinc	136	136	0.163	1.63	1.80	22.2	130	0.17
Omikviorok River								
Metals								
Aluminum	14,900	14,900	17.9	179	197	2,430	120	20
Arsenic	9.4	9.4	0.0113	0.113	0.124	1.53	10	0.15
Barium	484	484	0.581	5.81	6.39	78.9	21	3.8
Cadmium	0.59	0.59	0.000708	0.00708	0.00779	0.0961	1.5	0.064
Chromium	23.3	23.3	0.0280	0.280	0.308	3.80	0.86	4.4
Copper	15.6	15.6	0.0187	0.187	0.206	2.54	47	0.054
Fluoride	1.2	1.2	0.00144	0.0144	0.0158	0.196	7.8	0.025
Lead	19	19	0.0228	0.228	0.251	3.10	3.9	0.79
Manganese	2,140	2,140	2.57	25.7	28.2	349	980	0.36

Table 3-33. (cont.)

Analyte	Maximum Concentration Sediment (mg/kg dw)	Maximum Concentration Invertebrates (mg/kg dw)	Daily Exposure Sediment (mg/day)	Daily Exposure Invertebrates (mg/day)	Total Daily Intake	Body Weight Normalized Exposure	TRV (mg/kg-day)	Hazard Quotient
Mercury	0.06	0.06	0.0000720	0.000720	0.000792	0.00978	0.032	0.31
Molybdenum	0.66	0.66	0.000792	0.00792	0.00871	0.108	3.5	0.031
Nickel	57.3	57.3	0.0688	0.688	0.756	9.34	77	0.12
Selenium	0.7	0.7	0.000840	0.00840	0.00924	0.114	0.40	0.29
Thallium	0.141	0.141	0.000169	0.00169	0.00186	0.0230	0.24	0.097
Tin	5.7	5.7	0.00684	0.0684	0.0752	0.929	6.8	0.14
Vanadium	28.1	28.1	0.0337	0.337	0.371	4.58	11	0.42
Zinc	123	123	0.148	1.48	1.62	20.0	130	0.15
Anxiety Ridge Creek								
Metals								
Aluminum	8,310	8,310	9.97	99.7	110	1,354	120	11
Arsenic	11.4	11.4	0.0137	0.137	0.150	1.86	10	0.19
Barium	922	922	1.11	11.1	12.2	150	21	7.2
Cadmium	1.38	1.38	0.00166	0.0166	0.0182	0.225	1.5	0.15
Chromium	14.6	14.6	0.0175	0.175	0.193	2.38	0.86	2.8
Copper	20.1	20.1	0.0241	0.241	0.265	3.28	47	0.070
Fluoride	0.95	0.95	0.00114	0.0114	0.0125	0.155	7.8	0.020
Lead	142	142	0.170	1.70	1.87	23.1	3.9	5.9
Manganese	2,100	2,100	2.52	25.2	27.7	342	980	0.35
Mercury	0.089	0.089	0.000107	0.00107	0.00117	0.0145	0.032	0.45
Molybdenum	2.32	2.32	0.00278	0.0278	0.0306	0.378	3.5	0.11
Nickel	45.6	45.6	0.0547	0.547	0.602	7.43	77	0.10
Selenium	2.4	2.4	0.00288	0.0288	0.032	0.391	0.40	0.98
Thallium	0.322	0.322	0.000386	0.00386	0.00425	0.0525	0.24	0.22
Tin	1.1	1.1	0.00132	0.0132	0.0145	0.179	6.8	0.026
Vanadium	20.5	20.5	0.0246	0.246	0.271	3.34	11	0.30
Zinc	259	259	0.311	3.11	3.42	42.2	130	0.32
Reference Creeks								
Metals								
Aluminum	12,100	12,100	14.5	145	160	1,970	120	16
Arsenic	8.1	8.1	0.00972	0.0972	0.107	1.32	10	0.13
Barium	483	483	0.580	5.80	6.38	78.7	21	3.7
Cadmium	0.3	0.3	0.000360	0.00360	0.00396	0.0489	1.5	0.033
Chromium	19.9	19.9	0.0239	0.239	0.263	3.24	0.86	3.8
Copper	18.5	18.5	0.0222	0.222	0.244	3.01	47	0.064
Fluoride	1.2	1.2	0.00144	0.0144	0.0158	0.196	7.8	0.025
Lead	9.17	9.17	0.0110	0.110	0.121	1.49	3.9	0.38
Manganese	859	859	1.03	10.3	11.3	140	980	0.14
Mercury	0.04	0.04	0.0000480	0.000480	0.000528	0.00652	0.032	0.20
Molybdenum	0.52	0.52	0.000624	0.00624	0.00686	0.0847	3.5	0.024
Nickel	35	35	0.0420	0.420	0.462	5.70	77	0.074
Selenium	0.7	0.7	0.000840	0.00840	0.00924	0.114	0.40	0.29
Thallium	0.07	0.07	0.0000840	0.000840	0.000924	0.0114	0.24	0.048
Tin	2.4	2.4	0.00288	0.0288	0.0317	0.391	6.8	0.058
Vanadium	24.8	24.8	0.0298	0.298	0.327	4.04	11	0.37
Zinc	69.7	69.7	0.0836	0.836	0.920	11.4	130	0.087

Note: Hazard quotients greater than 1.0 are boxed.

Invertebrate data are modeled based on maximum sediment concentrations and are not measured values.

TRV - toxicity reference value

Table 3-34. Screening-level food web results for common snipe foraging in tundra ponds

Analyte	Maximum Concentration		Daily Exposure		Total Daily Intake	Body Weight Normalized Exposure	TRV (mg/kg-day)	Hazard Quotient
	Sediment (mg/kg dw)	Invertebrates (mg/kg dw)	Sediment (mg/day)	Invertebrates (mg/day)				
Site Tundra Ponds								
Metals								
Aluminum	5,270	5,270	6.32	63.2	69.6	859	120	7.2
Arsenic	8.7	8.7	0.0104	0.104	0.115	1.42	10	0.14
Barium	626	626	0.751	7.51	8.26	102	21	4.9
Cadmium	119	119	0.143	1.43	1.57	19.4	1.5	13
Chromium	15.3	15.3	0.0184	0.184	0.202	2.49	0.86	2.9
Copper	53.4	53.4	0.0641	0.641	0.705	8.70	47	0.19
Fluoride	4.4	4.4	0.00528	0.0528	0.0581	0.717	7.8	0.092
Lead	2,180	2,180	2.62	26.2	28.8	355	3.9	91
Manganese	745	745	0.894	8.94	9.83	121	980	0.12
Mercury	1.31	1.31	0.00157	0.0157	0.0173	0.213	0.032	6.7
Molybdenum	2.84	2.84	0.00341	0.0341	0.0375	0.463	3.5	0.13
Nickel	44.2	44.2	0.0530	0.530	0.583	7.20	77	0.094
Selenium	3.5	3.5	0.00420	0.0420	0.0462	0.570	0.40	1.4
Thallium	1.92	1.92	0.00230	0.0230	0.0253	0.313	0.24	1.3
Tin	41.2	41.2	0.0494	0.494	0.544	6.71	6.8	0.99
Vanadium	28.3	28.3	0.0340	0.340	0.374	4.61	11	0.42
Zinc	27,000	27,000	32.4	324	356	4,400	130	34
Reference Ponds								
Metals								
Aluminum	17,100	17,100	20.5	205	226	2,790	120	23
Arsenic	13	13	0.0156	0.156	0.172	2.12	10	0.21
Barium	772	772	0.926	9.26	10.2	126	21	6.0
Cadmium	0.66	0.66	0.000792	0.00792	0.00871	0.108	1.5	0.072
Chromium	28	28	0.0336	0.336	0.370	4.56	0.86	5.3
Copper	20.7	20.7	0.0248	0.248	0.273	3.37	47	0.072
Fluoride	1.2	1.2	0.00144	0.0144	0.0158	0.196	7.8	0.025
Lead	20.3	20.3	0.0244	0.244	0.268	3.31	3.9	0.85
Manganese	1,870	1,870	2.24	22.4	24.7	305	980	0.31
Mercury	0.07	0.07	0.0000840	0.000840	0.000924	0.0114	0.032	0.36
Molybdenum	1.35	1.35	0.00162	0.0162	0.0178	0.220	3.5	0.063
Nickel	70.3	70.3	0.0844	0.844	0.928	11.5	77	0.15
Selenium	3.1	3.10	0.00372	0.0372	0.0409	0.505	0.40	1.3
Thallium	0.174	0.174	0.000209	0.00209	0.00230	0.0284	0.24	0.12
Tin	6.3	6.3	0.00756	0.0756	0.0832	1.03	6.8	0.15
Vanadium	94.5	94.5	0.113	1.13	1.25	15.4	11	1.4
Zinc	138	138	0.166	1.66	1.82	22.5	130	0.17

Note: Hazard quotients greater than 1.0 are boxed.

Invertebrate data are modeled based on maximum sediment concentrations and are not measured values.

TRV - toxicity reference value

Table 3-35. Screening-level food-web results for black-bellied plover foraging in coastal lagoons

Analyte	Maximum Concentration		Daily Exposure		Total Daily Intake	Body Weight Normalized Exposure	TRV (mg/kg-day)	Hazard Quotient
	Sediment (mg/kg dw)	Invertebrates (mg/kg dw)	Sediment (mg/day)	Invertebrates (mg/day)				
Site Coastal Lagoons								
Metals								
Aluminum	15,300	15,300	115	398	513	2,660	120	22
Arsenic	17.9	17.9	0.135	0.465	0.600	3.11	10	0.31
Barium	350	350	2.64	9.09	11.7	60.8	21	2.9
Cadmium	8.1	8.1	0.0610	0.210	0.272	1.41	1.5	0.94
Chromium	27.2	27.2	0.205	0.707	0.912	4.72	0.86	5.5
Copper	28.2	28.2	0.212	0.733	0.945	4.90	47	0.10
Fluoride	8.6	8.6	0.0648	0.223	0.288	1.49	7.8	0.19
Lead	302	302	2.28	7.85	10.1	52.4	3.9	13
Manganese	274	274	2.06	7.12	9.18	47.6	980	0.049
Mercury	0.096	0.096	0.000723	0.00249	0.00322	0.0167	0.032	0.52
Molybdenum	3.39	3.39	0.0255	0.0881	0.114	0.589	3.5	0.17
Nickel	39	39	0.294	1.01	1.31	6.77	77	0.088
Selenium	2.2	2.2	0.0166	0.0572	0.0737	0.382	0.40	0.96
Thallium	0.184	0.184	0.00139	0.00478	0.00617	0.0320	0.24	0.13
Tin	6.7	6.7	0.0505	0.174	0.225	1.16	6.8	0.17
Vanadium	35.1	35.1	0.264	0.912	1.18	6.10	11	0.55
Zinc	1,590	1,590	12.0	41.3	53.3	276	130	2.1
Reference Lagoons								
Metals								
Aluminum	14,800	14,800	112	385	496	2,570	120	21
Arsenic	4.9	4.9	0.0369	0.127	0.164	0.851	10	0.085
Barium	271	271	2.04	7.04	9.08	47.1	21	2.2
Cadmium	0.49	0.49	0.00369	0.0127	0.0164	0.0851	1.5	0.057
Chromium	24.9	24.9	0.188	0.647	0.835	4.32	0.86	5.0
Copper	18.7	18.7	0.141	0.486	0.627	3.25	47	0.069
Fluoride	1.2	1.2	0.00904	0.0312	0.0402	0.208	7.8	0.027
Lead	23	23	0.173	0.598	0.771	3.99	3.9	1.0
Manganese	129	129	0.972	3.35	4.32	22.4	980	0.023
Mercury	0.06	0.06	0.000452	0.00156	0.00201	0.0104	0.032	0.33
Molybdenum	0.98	0.98	0.00738	0.0255	0.0328	0.170	3.5	0.049
Nickel	37	37	0.279	0.961	1.24	6.43	77	0.083
Selenium	1.4	1.4	0.0105	0.0364	0.0469	0.243	0.40	0.61
Thallium	0.103	0.103	0.000776	0.00268	0.00345	0.0179	0.24	0.075
Tin	5.1	5.1	0.0384	0.133	0.171	0.886	6.8	0.13
Vanadium	31.5	31.5	0.237	0.818	1.06	5.47	11	0.50
Zinc	371	371	2.79	9.63	12.4	64.4	130	0.50

Note: Hazard quotients greater than 1.0 are boxed.

Invertebrate data are modeled based on maximum sediment concentrations and are not measured values.

TRV - toxicity reference value

Table 3-36. Results of screening against lowest ecological screening benchmarks

	Environment								
	Terrestrial	Streams		Ponds		Lagoons		Marine	
Chemical	Tundra Soil	Sediment	Water	Sediment	Water	Sediment	Water	Sediment	Water
Aluminum	Fail	Pass	Fail	Pass	Fail	NB	NB	NB	NB
Antimony	Fail	NB	NB	NB	NB	NB	NB	NB	NB
Arsenic	Fail	Fail	ND[a]	Pass	Pass	Fail	Fail	Fail	Pass
Barium	Fail	NB	NB	NB	NB	NB	NB	NB	NB
Cadmium	Fail	Fail	Fail	Fail	Fail	Fail	Pass	Fail	Pass
Chromium	Fail	Pass	ND[a]	Pass	Pass	Pass	Pass	Pass	ND[a]
Cobalt	Fail	NB	NB	NB	NB	NB	NB	NB	NB
Copper	Fail	Pass	Pass	Fail	Fail	Pass	Pass	Fail	Fail
Fluoride	Pass	ND[b]	NB	NB	NB	NB	NB	NB	NB
Iron	Fail	Pass	Fail	Pass	Fail	NB	NB	NB	NB
Lead	Fail	Fail	Fail	Fail	Fail	Fail	Pass	Fail	Pass
Manganese	Fail	Pass	NB	Pass	NB	NB	NB	NB	NB
Mercury	Fail	Pass	ND[a]	Fail	ND[a]	Pass	ND[a]	Fail	ND[a]
Molybdenum	Fail	NB	NB	NB	NB	NB	NB	NB	NB
Nickel	Fail	Fail	Pass	Fail	Pass	Fail	Fail	Fail	ND[c]
Selenium	Fail	NB	Pass	NB	ND[a]	NB	Pass	NB	Pass
Silver	Fail	NB	ND[a]	NB	ND[a]	Pass	Pass	Fail	Pass
Strontium	NB	NB	NB	NB	NB	NB	NB	NB	NB
Thallium	Fail	NB	NB	NB	NB	NB	NB	NB	NB
Tin	Pass	NB	NB	NB	NB	NB	NB	NB	NB
Vanadium	Fail	NB	NB	NB	NB	NB	NB	NB	NB
Zinc	Fail	Fail	Fail	Fail	Fail	Fail	Fail	Fail	ND[a]

Note: Fail - maximum detected concentration exceeds the lowest benchmark
NB - no benchmark
ND - undetected in all samples
Pass - maximum detected concentration is below the lowest benchmark

[a] Maximum value expressed as one-half of the detection limit is below the screening benchmark.

[b] No benchmark.

[c] Maximum value expressed as one-half of the detection limit is above the screening benchmark.

Table 3-37. Results of screening against the marine sediment quality standards and no-effect concentrations[a]

	Environment								
	Terrestrial	Streams		Ponds		Lagoons		Marine	
Chemical	Tundra Soil	Sediment	Water	Sediment	Water	Sediment	Water	Sediment	Water
Aluminum	Fail	--	Fail	--	Fail	NB	NB	NB	NB
Antimony	Fail	NB	NB	NB	NB	NB	NB	NB	NB
Arsenic	Fail	Pass	--	--	--	Pass	Fail	Pass	--
Barium	Fail	NB	NB	NB	NB	NB	NB	NB	NB
Cadmium	Fail	Pass	Fail	Fail	Fail	Fail	--	Fail	--
Chromium	Fail	--	--	--	--	--	--	--	--
Cobalt	Fail	NB	NB	NB	NB	NB	NB	NB	NB
Copper	Fail	--	--	Pass	Fail	--	--	Pass	Fail
Fluoride	--	ND[b]	NB	NB	NB	NB	NB	NB	NB
Iron	Fail	--	Fail	--	Fail	NB	NB	NB	NB
Lead	Fail	Fail	Fail	Fail	Fail	Pass	--	Fail	--
Manganese	Fail	--	NB	--	NB	NB	NB	NB	NB
Mercury	Fail	--	--	Fail[c]	--	--	--	Fail	--
Molybdenum	Fail	NB	NB	NB	NB	NB	NB	NB	NB
Nickel	Fail	Fail	--	Fail	--	Pass[d]	Fail	Pass[d]	ND[e]
Selenium	Fail	NB	--	NB	--	NB	--	NB	--
Silver	Fail	NB	--	NB	--	--	--	Pass	--
Strontium	NB	NB	NB	NB	NB	NB	NB	NB	NB
Thallium	Fail	NB	NB	NB	NB	NB	NB	NB	NB
Tin	--	NB	NB	NB	NB	NB	NB	NB	NB
Vanadium	Fail	NB	NB	NB	NB	NB	NB	NB	NB
Zinc	Fail	Pass	Fail	Fail	Fail	Fail	Fail	Fail	--

Note: -- - chemical passed earlier screening tier
Fail - maximum detected concentration exceeds the lowest benchmark
NB - no benchmark
ND - undetected in all samples
Pass - maximum detected concentration is below the lowest benchmark

[a] The marine sediment quality standard values are used for marine sediment comparison, and the no-effect concentration values are used for freshwater sediment comparison in this table. There is no change with regard to water screening in comparison to the previous screening table.

[b] No benchmark.

[c] Result of comparison with the probable effects concentration (MacDonald et al. 2000); the NEC was not derived for mercury (Ingersoll et al. 1996).

[d] Result of comparison with the effects range median (Long et al. 1995); the SQS was not derived for nickel.

[e] Maximum value expressed as one-half of the detection limit is above the screening benchmark.

Table 3-38. Results of screening based on frequency of ecological benchmark exceedances

	Environment								
	Terrestrial	Streams		Ponds		Lagoons		Marine	
Chemical	Tundra Soil	Sediment	Water	Sediment	Water	Sediment	Water	Sediment	Water
Aluminum	Fail	--	Fail	--	Fail	NB	NB	NB	NB
Antimony	Fail	NB	NB	NB	NB	NB	NB	NB	NB
Arsenic	Fail	--	--	--	--	--	Fail	--	--
Barium	Fail	NB	NB	NB	NB	NB	NB	NB	NB
Cadmium	Fail	--	Pass	Fail	Fail	Pass	--	Pass	--
Chromium	Fail	--	--	--	--	--	--	--	--
Cobalt	Fail	NB	NB	NB	NB	NB	NB	NB	NB
Copper	Fail	--	--	--	Fail	--	--	--	Pass
Fluoride	--	ND[a]	NB	NB	NB	NB	NB	NB	NB
Iron	Fail	--	Pass	--	Fail	NB	NB	NB	NB
Lead	Fail	Pass	Pass	Fail	Fail	--	--	Pass	--
Manganese	Fail	--	NB	--	NB	NB	NB	NB	NB
Mercury	Fail	--	--	Fail[b]	--	--	--	Pass	--
Molybdenum	Fail	NB	NB	NB	NB	NB	NB	NB	NB
Nickel	Fail	Fail	--	Fail	--	--	Fail	--	ND[c]
Selenium	Fail	NB	--	NB	--	NB	--	NB	--
Silver	Fail	NB	--	NB	--	--	--	--	--
Strontium	NB	NB	NB	NB	NB	NB	NB	NB	NB
Thallium	Fail	NB	NB	NB	NB	NB	NB	NB	NB
Tin	--	NB	NB	NB	NB	NB	NB	NB	NB
Vanadium	Fail	NB	NB	NB	NB	NB	NB	NB	NB
Zinc	Fail	--	Pass	Fail	Fail	Fail	Pass	Pass	--

Note: -- - chemical passed earlier screening tier
Fail - maximum detected concentration exceeds the benchmark in at least 10 percent of samples analyzed
NB - no benchmark
ND - undetected in all samples
Pass - maximum detected concentration exceeds the benchmark in less than 10 percent of samples analyzed

[a] No benchmark.

[b] Result of comparison with the probable effects concentration (MacDonald et al. 2000); the NEC was not derived for mercury (Ingersoll et al. 1996).

[c] Maximum value expressed as one-half of the detection limit is above the screening benchmark in at least 10 percent of samples analyzed.

Table 3-39. Results of statistical comparison with reference data

	Environment								
	Terrestrial	Streams		Ponds		Lagoons		Marine	
Chemical	Tundra Soil	Sediment	Water	Sediment	Water	Sediment	Water	Sediment	Water
Aluminum	Pass	--	Pass	--	NA	Pass	Pass	Pass	Pass
Antimony	NA	Fail	NA	Fail	NA	Pass	Fail	NA	Pass
Arsenic	Fail	--	--	--	--	--	Pass	--	--
Barium	NA	Pass	Pass	Pass	Pass	Pass	Pass	Fail	Pass
Cadmium	Fail	--	--	Fail	NA	--	--	--	--
Chromium	Pass	--	--	--	--	--	--	--	--
Cobalt	Pass	Fail	Pass	NA	NA	Pass	Pass	Pass	Pass
Copper	Fail	--	--	--	Pass	--	--	--	--
Fluoride	--	NA	Fail	NA	Pass	NA	Fail	Pass	Pass
Iron	Pass	--	--	--	Pass	Pass	Pass	Pass	Pass
Lead	Fail	--	--	NA	Pass	--	--	--	--
Manganese	Pass	--	Pass	--	NA	Fail	Pass	Pass	Pass
Mercury	NA	--	--	NA	--	--	--	--	--
Molybdenum	Fail	Fail	Fail	Fail	Pass	Pass	Fail	Pass	Pass
Nickel	Pass	Fail	--	Pass	--	--	Pass	--	NA
Selenium	Fail	Fail	--	Pass	--	Pass	--	NA	--
Silver	NA	Fail	--	NA	--	--	--	--	--
Strontium	Fail	Fail	Pass	Pass	Pass	Pass	Pass	Fail	Pass
Thallium	NA	Fail	NA	Pass	NA	Pass	Pass	Pass	NA
Tin	--	Pass	NA	NA	NA	NA	NA	NA	NA
Vanadium	NA	Pass	NA	Pass	Pass	Pass	Pass	Pass	NA
Zinc	Fail	--	--	Fail	NA	Fail	--	--	--

Note: -- - chemical passed earlier screening tier
Fail - site concentrations significantly greater than reference concentrations
NA - not applicable; no statistical comparison was made because of high frequency of nondetects; or the confidence interval for the site mean straddles zero as a result of small sample size or high variability
Pass - site concentrations not significantly greater than reference concentrations

Table 3-40. Chemicals of potential concern retained for ecological risk analysis

	Environment								
	Terrestrial	Streams		Ponds		Lagoons		Marine	
Chemical	Tundra Soil	Sediment	Water	Sediment	Water	Sediment	Water	Sediment	Water
Aluminum	--	--	--	--	Fail	--	--	--	--
Antimony	Fail	NB	NB	NB	NB	--	NB	NB	--
Arsenic	Fail	--	--	--	--	--	--	--	--
Barium	Fail	--	--	--	--	--	--	NB	--
Cadmium	Fail	--	--	Fail	Fail	--	--	--	--
Chromium	--	--	--	--	--	--	--	--	--
Cobalt	--	NB	--	NB	NB	--	--	--	--
Copper	Fail	--	--	--	--	--	--	--	--
Fluoride	--	ND[a]	NB	NB	--	NB	NB	--	--
Iron	--	--	--	--	--	--	--	--	--
Lead	Fail	--	--	Fail	--	--	--	--	--
Manganese	--	--	--	--	NB	NB	--	--	--
Mercury	Fail	--	--	Fail	--	--	--	--	--
Molybdenum	Fail	NB	NB	NB	--	--	NB	--	--
Nickel	--	Fail	--	--	--	--	--	--	ND[b]
Selenium	Fail	NB	--	--	--	--	--	NB	--
Silver	Fail	NB	--	NB	--	--	--	--	--
Strontium	NB	NB	--	--	--	--	--	NB	--
Thallium	Fail	NB	NB	--	NB	--	--	--	NB
Tin	--	--	NB	NB	NB	NB	NB	NB	NB
Vanadium	Fail	--	NB	--	--	--	--	--	NB
Zinc	Fail	--	--	Fail	Fail	Fail	--	--	--

Note: -- - chemical eliminated from further evaluation
Fail - chemical retained as a CoPC for the baseline ERA
NB - no benchmark; chemical retained as a CoPC for the baseline ERA
ND - undetected in all samples; chemical retained as a CoPC for the baseline ERA

[a] No benchmark.

[b] Maximum value expressed as one-half of the detection limit is above the screening benchmark in at least 10 percent of samples analyzed.

Table 4-1. EPA IEUBK lead model exposure parameters and input values

Parameter	Input Value(s)	Source
Air		
Outdoor lead concentration ($\mu g/m^3$)	0.100	EPA default
Indoor air lead concentration (percent of outdoor air)	30 percent	EPA default
Time spent outdoors (hours/day)	1, 2, 3, 4, 4, 4,4[a]	EPA default
Ventilation rates (m^3/day)	2, 3, 5, 5, 5, 7 ,7[a]	EPA default
Lung absorption (percentage)	32 percent	EPA default
Diet		
Diet intake (μg/day)	To be determined	Site data
Alternative diet values	To be determined	Site data
Gastrointestinal absorption from diet (percent)	50 percent	EPA default
Drinking Water		
Lead concentration in drinking water (μg/L)	0.7	Site data
Drinking water intake (L/day)	0.20, 0.50, 0.52, 0.53, 0.55, 0.58, 0.59[a]	EPA default
Alternative water values	Not used	EPA default
GI absorption from drinking water (percent)	50	EPA default
Soil/Dust		
Soil lead levels (ppm; $\mu g/g$)	To be determined	Site data
Indoor dust lead levels (percent of soil levels)	70 percent	EPA default
Ingestion weighting factor (percent soil/percent dust)	45/55	EPA default
Amount of soil/dust ingested daily (g/day)	0.085, 0.135, 0.135, 0.135, 0.100, 0.090, 0.085[a]	EPA default
GI absorption from soil and dust (percent)	30, 9.7	DPH (2001), Arnold et al. (2003)
Other		
Paint lead intake (μg/day)	0.0	EPA default
Alternative paint values	Not used	EPA default
GI absorption from leaded paint (percent)	30 percent	EPA default
Maternal contribution method	Infant model	EPA default
Maternal blood lead at birth of child (μg/dL)	2.5	EPA default
Geometric standard deviation	1.6	EPA default

[a] Value varies by age group. Values listed are for the following ages, respectively: 0–1, 1–2, 2–3, 3–4, 4–5, 5–6, and 6–7.

Table 4-2. Bioavailability of lead in Red Dog ore concentrate

Lead Concentration in Soil (mg/kg)	Blood Lead (μg/dL)		Relative Bioavailability	Child Absolute Bioavailability	Adult Absolute Bioavailability
	Lead Acetate	Red Dog Concentrate			
0		5.05	--	--	--
10	16	4.32	27.0%	13.5%	5.4%
30	31.8	5.65	17.8%	8.9%	3.6%
100	84.8	11.5	13.6%	6.8%	2.7%
Average			19.4%	9.7%	3.9%

Source: Arnold and Middaugh (2001)

Table 4-3. Estimated subsistence food consumption

	Mean per Capita Consumption (g/day)			Caloric Intake Weighted Mean per Capita Consumption (g/day)	
	Kivalina	Noatak	Average User	Adult	Child
Land Mammals	212.1	305.8	259.0	168	84
Caribou	177.5	300.6	239.1	155	78
Moose	70.0	36.9	53.4	35	17
Migratory Birds	10.6	9.9	10.3	6.7	3.3
Game Birds	3.1	3.1	3.1	2.0	1.0
Ptarmigan	3.1	3.1	3.1	2.0	1.0
All Fish	314.8	248.7	281.7	183	91
Salmon	29.2	216.1	122.6	80	40
Non-salmon fish	296.4	85.0	190.7	124	62
Char	252.3	57.7	155.0	101	50
White fish	28.2	36.0	32.1	21	10
Cod	24.8	1.1	12.9	8.4	4.2
Marine Invertebrates	1.8	3.8	2.8	1.8	0.9
Clams	0.0	1.3	0.6	0.4	0.2
Crabs	0.8	6.4	3.6	2.3	1.2
Shrimp	1.6	0.0	0.8	0.5	0.3
Marine Mammals	415.1	106.0	260.6	169	85
Seal	251.8	101.6	176.7	115	57
Walrus	101.1	52.9	77.0	50	25
Whale	89.8	20.2	55.0	36	18
Vegetation	18.3	7.5	12.9	8.4	4.2
Berries	17.5	8.2	12.9	8.4	4.2
Plants/greens/mushrooms	1.5	2.5	2.0	1.3	0.7
Sum of Main Categories	976	685	830	539	270
Total kcal/day (@5.1 kcal/g)	4,977	3,492	4,234	2,750	1,375
Caloric Intake Weighting Factor	--	--	--	0.65	0.32

Note: Data from Community Profile Database (DFG 2001). Kivalina data are from 1992. Noatak data are from 1994.

The sum of consumption rates for individual food items, or for sub-categories within a category, does not equal the consumption rate for the entire category in the database. For example, the sum of salmon and non-salmon fish consumption does not equal all fish consumption. This could be an artifact of the statistical methods used to derive consumption rates for entire categories based on data for individual items.

Table 4-4. Daily dietary intake of Alaska native adults

	Males		Females	
	grams	kcal	grams	kcal
Protein	127	508	90	360
Fat	117	1,053	81	729
Carbohydrates	282	1,128	214	856
Total Energy	526	2,689	385	1,945
Average kcal/g		5.1		5.1

Source: Nobmann et al. (1992)

Note: kcal - kilocalories; commonly called calories. Caloric intake was calculated by multiplying the intake in grams from Nobmann et al. (1992) by the number of kcal/g in each energy source: protein, 4 kcal/g; fat, 9 kcal/g; carbohydrate, 4 kcal/g.

Table 5-1. Refined assessment endpoints, representative receptors, and measurement endpoints

Environment	Assessment Endpoint	Representative Receptor[a]	Measurement Endpoint
Tundra	Structure and function of terrestrial plant communities	Terrestrial plant communities	Plant abundance, diversity, biomass, percent cover
Tundra	Structure and function of tundra soil fauna communities	Tundra soil fauna communities	Not directly assessed, evaluated through terrestrial plant community analysis
Tundra	Survival, growth, and reproduction of terrestrial avian herbivore populations	Willow ptarmigan	Range of modeled total dietary exposures (based on measured CoPC concentrations in food, soil, and surface water) relative to avian TRVs
Tundra	Survival, growth, and reproduction of terrestrial mammalian herbivore populations	Tundra vole; caribou; moose	Range of modeled total dietary exposures (based on measured CoPC concentrations in food, soil, and surface water) relative to mammalian TRVs
Tundra	Survival, growth, and reproduction of terrestrial avian invertivore populations	Lapland longspur	Range of modeled total dietary exposures (based on measured CoPC concentrations in food, soil, and surface water) relative to avian TRVs
Tundra	Survival, growth, and reproduction of terrestrial mammalian invertivore populations	Tundra shrew	Range of modeled total dietary exposures (based on measured CoPC concentrations in food, soil, and surface water) relative to mammalian TRVs
Tundra	Survival, growth, and reproduction of terrestrial avian carnivore populations	Snowy owl	Range of modeled total dietary exposures (based on measured CoPC concentrations in food, soil, and surface water) relative to avian TRVs
Tundra	Survival, growth, and reproduction of terrestrial mammalian carnivore populations	Arctic fox	Range of modeled total dietary exposures (based on measured CoPC concentrations in food, soil, and surface water) relative to mammalian TRVs
Streams	Structure and function of stream aquatic and wetland plant communities	Stream aquatic and wetland plant communities	Plant abundance, diversity, biomass, percent cover
Streams	Structure and function of stream aquatic invertebrate communities	Stream aquatic invertebrate communities	Abundance and diversity of stream aquatic invertebrates
Streams	Survival, growth, and reproduction of stream avian herbivore populations	Green-winged teal	Range of modeled total dietary exposures (based on measured CoPC concentrations in food, sediment, and surface water) relative to avian TRVs
Streams	Survival, growth, and reproduction of stream mammalian herbivore populations	Muskrat	Range of modeled total dietary exposures (based on measured CoPC concentrations in food, sediment, and surface water) relative to mammalian TRVs

Table 5-1. (cont.)

Environment	Assessment Endpoint	Representative Receptor[a]	Measurement Endpoint
Streams	Survival, growth, and reproduction of stream avian invertivore populations	Common snipe	Range of modeled total dietary exposures (based on measured CoPC concentrations in food, sediment, and surface water) relative to avian TRVs
Tundra ponds	Structure and function of tundra pond aquatic and wetland plant communities	Tundra pond aquatic and wetland plant communities	Plant abundance, diversity, biomass, percent cover
Tundra ponds	Structure and function of tundra pond aquatic invertebrate communities	Tundra pond aquatic invertebrate communities	Abundance and diversity of tundra pond aquatic invertebrates
Tundra ponds	Survival, growth, and reproduction of tundra pond avian herbivore populations	Green-winged teal	Range of modeled total dietary exposures (based on measured CoPC concentrations in food, sediment, and surface water) relative to avian TRVs
Tundra ponds	Survival, growth, and reproduction of tundra pond mammalian herbivore populations	Muskrat	Range of modeled total dietary exposures (based on measured CoPC concentrations in food, sediment, and surface water) relative to mammalian TRVs
Tundra ponds	Survival, growth, and reproduction of tundra pond avian invertivore populations	Common snipe	Range of modeled total dietary exposures (based on measured CoPC concentrations in food, sediment, and surface water) relative to avian TRVs
Coastal lagoons	Structure and function of coastal lagoon aquatic and wetland plant communities	Coastal lagoon aquatic and wetland plant communities	Plant abundance, diversity, biomass, percent cover
Coastal lagoons	Structure and function of coastal lagoon aquatic invertebrate communities	Coastal lagoon aquatic invertebrate communities	Abundance and diversity of coastal lagoon aquatic invertebrates
Coastal lagoons	Survival, growth, and reproduction of coastal lagoon avian herbivore populations	Brant	Range of modeled total dietary exposures (based on measured CoPC concentrations in food and sediment) relative to avian TRVs
Coastal lagoons	Survival, growth, and reproduction of coastal lagoon avian invertivore populations	Black-bellied plover	Range of modeled total dietary exposures (based on measured CoPC concentrations in food and sediment) relative to avian TRVs

Note: CoPC - chemical of potential concern
TRV - toxicity reference value

[a] Receptors to be evaluated in the risk assessment.

Table 5-2. Parameters proposed for use in the deterministic exposure analysis

Representative Receptor	Community	Body Weight (kg)	Food Ingestion Rate (kg/day(dry wt)	Soil/Sediment Ingestion Rate (kg/day dry wt)	Water Ingestion Rate (L/day)[a]	Diet Composition (percent)		Time Use (days)	
Terrestrial									
Willow ptarmigan	Terrestrial avian herbivores	0.53 [b]	0.060 [c]	0.0056 [d]	0.038	90% shrubs, 10% herbaceous plants	[e]	365	[f]
Tundra vole	Terrestrial mammalian herbivores	0.047 [g]	0.0085 [h]	0.00020 [i]	0.0063	90% herbaceous plants, 5% moss, 5% lichen	[j]	365	[f]
Caribou	Terrestrial mammalian herbivores	107 [k]	5.0 [l]	0.34 [m]	6.6	70% lichen, 10% shrubs, 10% herbaceous plants, 10% moss	[n]	150	[o]
Moose	Terrestrial mammalian herbivores	339 [p]	2.6 [q]	0.052 [m]	19	90% shrubs, 10% herbaceous plants	[r]	365	[f]
Lapland longspur	Terrestrial avian invertevores	0.0254 [s]	0.0053 [t]	0.000074 [u]	0.0050	90% invertebrates, 10% herbaceous plants	[v]	150	[w]
Tundra shrew	Terrestrial mammalian invertevores	0.0064 [x]	0.0021 [y]	0.00011 [z]	0.0011	100% invertebrates	[aa]	365	[f]
Snowy owl	Terrestrial avian carnivores	2.28 [bb]	0.10 [cc]	0.0020 [dd]	0.10	100% small mammals	[ee]	365	[f]
Arctic fox	Terrestrial mammalian carnivores	3.2 [ff]	0.11 [gg]	0.0031 [hh]	0.28	100% small mammals	[ii]	365	[f]
Freshwater Aquatic									
Green-winged teal	Freshwater aquatic avian herbivores	0.32 [jj]	0.053 [kk]	0.0010 [ll]	0.027	85% herbaceous plants, 15% invertebrates	[mm]	123	[nn]
Muskrat	Freshwater aquatic mammalian herbivores	0.932 [oo]	0.070 [pp]	0.0014 [dd]	0.093	100% herbaceous plants	[qq]	365	[f]
Common snipe	Freshwater aquatic avian invertevores	0.116 [jj]	0.015 [rr]	0.0016 [ss]	0.014	90% invertebrates, 10% herbaceous plants	[tt]	109	[uu]
Coastal Lagoon									
Brant	Marine avian herbivores	1.23 [jj]	0.13 [kk]	0.011 [vv]	0.068	95% herbaceous plants, 5% moss	[ww]	126	[xx]
Black-bellied plover	Marine avian invertevores	0.214 [yy]	0.028 [zz]	0.0082 [aaa]	0.021	100% invertebrates	[bbb]	124	[ccc]

[a] Based on U.S. EPA (1993) drinking water ingestion equations for all birds or all mammals.

[b] Mean female body weight from West et al. (1970).

[c] Estimated from Andreev (1991).

[d] Based on 9.3 percent soil in wild turkey diet from Beyer et al. (1994).

[e] Estimated from diets reported for Alaska in Hannon et al. (1998).

[f] Assumes receptor is present year-round at the site.

[g] Mean female body weight from Bee and Hall (1956).

[h] Based on Nagy et al. (1999) allometric equation for Rodentia.

[i] Based on 2.4 percent soil in meadow vole diet from Beyer et al. (1994).

[j] Estimated from summer and winter diets at Pearce Point, NWT (Bergman and Krebs 1993).

Table 5-2. (cont.)

[k] Mean female in Alaska from Silva and Downing (1995).

[l] Based on mean value from Hanson et al. (1975).

[m] Based on 6.8 percent soil in bison diet from Beyer et al. (1994).

[n] Based on diets reported in Miller (1976), Boertje (1990), and Scotter (1967).

[o] Best professional judgement based on Lent (1966), Hemming (1987, 1988, 1989, 1991), Pollard (1994a,b).

[p] Mean body weight for female Alaskan moose measured at the Kenai Moose Research Center, Soldotna, AK (Franzmann et al. 1978).

[q] Based on Nagy et al. (1999) allometric equation for herbivores.

[r] Estimated from diets reported for Alaska in Franzmann and Schwartz (1997)

[s] Mean female body weight from Irving (1960).

[t] Calculated using an average female daily energy budget of 118 KJ/day and average prey caloric value of 22.16 KJ/g from Custer et al. (1986).

[u] Based on 1.4 percent soil in Lapland longspur diet reported by URS Team (1996).

[v] Estimated from summer diets near Barrow, AK (Custer and Pitelka 1978).

[w] Based on 150 days from first to last sighting in Cape Thompson area reported by Williamson et al. (1966).

[x] Mean body weight from Bee and Hall (1956) and Martell and Pearson (1978).

[y] Based on measured food consumption from Buckner (1964), assuming a mid-range moisture content of 75 percent in invertebrates from U.S. EPA (1993).

[z] Best professional judgement based on Beyer et al. (1994).

[aa] Based on Yudin (1962, as cited in Aitchison 1987 and Buckner 1964).

[bb] Mean female body weight from Kerlinger and Lein (1988).

[cc] Estimated from Gessaman (1972) and Pitelka et al. (1955) assuming a moisture content of 68 percent in diet from U.S. EPA (1993).

[dd] Based on minimum soil ingestion rate from Beyer et al. (1994).

[ee] Simplified from Parmelee (1992).

[ff] Mean female body weight from Anthony (1997).

[gg] Based on Nagy et al. (1999) allometric equation for Carnivora.

[hh] Based on 2.8 percent soil in red fox diet from Beyer et al. (1994).

[ii] Simplified from Anthony et al. (2000).

[jj] Mean female body weight from Dunning (1993).

[kk] Based on Nagy et al. (1999) allometric equation for all birds.

[ll] Based on 1.9 percent sediment in green-winged teal diet from Beyer et al. (1999).

[mm] Estimated from autumn diet in southeastern Alaska (Hughes and Young 1982).

[nn] Based on 123 days from first to last sighting in Cape Thompson area reported by Williamson et al. (1966).

[oo] Mean body weight from Fuller (1951).

Table 5-2. (cont.)

[pp] Estimated from Campbell et al. (1998).

[qq] Based on diets reported in U.S. EPA (1993).

[rr] Based on Nagy et al. (1999) allometric equation for Insectivores.

[ss] Based on 10.4 percent soil in American woodcock diet from Beyer et al. (1994).

[tt] Based on diets reported in Mueller (1999).

[uu] Based on 109 days from first to last sighting in Cape Thompson area reported by Williamson et al. (1966).

[vv] Based on 8.2 percent soil in Canada goose diet from Beyer et al. (1994).

[ww] Based on breeding season diets reported in Reed et al. (1998).

[xx] Based on 126 days from first to last sighting in Cape Thompson area reported by Williamson et al. (1966).

[yy] Mean female body weight for Alaska from Paulson (1995).

[zz] Based on Nagy et al. (1999) allometric equation for Charadriiformes.

[aaa] Based on 29% sediment in black-bellied plover diet from Hui and Beyer (1998).

[bbb] Based on breeding season diets reported in Paulson (1995).

[ccc] Based on 124 days from first to last sighting of American golden plover in Cape Thompson area reported by Williamson et al. (1966).

Table 5-3. Parameters proposed for use with probabilistic exposure analysis for willow ptarmigan

	Distribution	Value
Receptor Population Parameters		
Body mass (g)	Normal	Mean 530; SD 36[a]
Food ingestion rate (g dry weight/day)	Functional	Mean 24.89, SD 1.45[b]
Proportion of diet represented by shrubs (percent)	Uniform	Min 85%; Max 100%[c]
Proportion of diet represented by herbaceous plants (percent)	Functional	100% - %shrubs[c]
Soil ingestion rate (percent of food ingestion rate)	Uniform	Min 2%, Max 9.3 %[d]
Water ingestion rate (mL/day)	Functional	Mean 38.5; SD 1.6[e]
Regional Habitat Utilization Parameters		
Home range (ha)	Uniform	Mean 3.9; SD 2.8[f]
Temporal Migration Parameters		
Time use (days)	Absolute value	365[g]

Note: SD - standard deviation

[a] Mean female body weight from West et al. (1970).

[b] Based on Nagy et al. (1999) allometric equation for Galliformes.

[c] Estimated from diets reported for Alaska in Hannon et al. (1998).

[d] Based on a 9.3 percent soil in wild turkey diet from Beyer et al. (1994).

[e] Based on U.S. EPA (1993) drinking water ingestion equations for all birds.

[f] Based on mean territory size of monogamous males from Hannon et al. (1998).

[g] Assumes receptor is present year-round at the site.

Table 5-4. Parameters proposed for use with probabilistic exposure analysis for tundra vole

	Distribution	Value
Receptor Population Parameters		
Body mass (g)	Triangular	Mean 47; range 29–65[a]
Food ingestion rate (g dry weight/day)	Functional	Mean 8.5; SD 0.94[b]
Proportion of diet represented by terrestrial herbaceous vegetation (percent)	Uniform	Min 80%; Max 90 %[c]
Proportion of diet represented by lichen (percent)	Functional	100% - % herbaceous/2[c]
Proportion of diet represented by moss (percent)	Functional	100% - % herbaceous/2[c]
Soil ingestion rate (percent of food ingestion rate)	Uniform	Min 0 %; Max 2.4%[d]
Water ingestion rate (mL/day)	Functional	Mean 6.3; SD 0.88[e]
Regional Habitat Utilization Parameters		
Home range (m^2)	Uniform	Min 366; Max 1,087[f]
Temporal Migration Parameters		
Time use (days/year)	Absolute value	365[g]

Note: SD - standard deviation

[a] Mean and range of female body weights from Bee and Hall (1956).

[b] Based on Nagy et al. (1999) allometric equation for Rodentia.

[c] Estimated from summer and winter diets at Pearce Point, NWT (Bergman and Krebs 1993).

[d] Based on 2.4 percent soil in meadow vole diet from Beyer et al. (1994).

[e] Based on U.S. EPA (1993) drinking water ingestion equations for all mammals.

[f] Based on mean home ranges for female tundra voles at Pearce Pt., NWT (Lambin et al. 1992) and female singing voles (*Microtus* miurus) near Toolik Lake, AK (Batzli and Henttonen 1993).

[g] Assumes receptor is present year-round at the site.

Table 5-5. Parameters proposed for use with probabilistic exposure analysis for caribou

	Distribution	Value
Receptor Population Parameters		
Body mass (kg)	Triangular	Mean 107; range 80–120[a]
Food ingestion rate (kg dry weight/day)	Triangular	Mean 5.0; range 3.7–6.9[b]
Proportion of diet represented by lichen (percent)	Uniform	50–80[c]
Proportion of diet represented by shrubs (percent)	Functional	100% - % herbaceous/3
Proportion of diet represented by herbaceous plants (percent)	Functional	100% - % herbaceous/3
Proportion of diet represented by moss (percent)	Functional	100% - % herbaceous/3[c]
Soil ingestion rate (percent of food ingestion rate)	Uniform	Min 0 %; Max 6.8%[d]
Water ingestion rate (L/kg body weight)	Functional	Mean 6.33; SD 0.46[e]
Regional Habitat Utilization Parameters		
Home range (km)	NA	NA
Temporal Migration Parameters		
Time use (days)	Triangular	Mean 150; range 30–270[f]

Note: NA - not applicable
SD - standard deviation

[a] Mean female body weight in Alaska from Silva and Downing (1995); range of female body weights from DFG (2001b).

[b] Based on Hanson et al. (1975).

[c] Estmated from caribou in Northwest Territories, Canada from Scotter (1967).

[d] Based on 6.8 percent soil in bison diet from Beyer et al. (1994).

[e] Based on U.S. EPA (1993) drinking water ingestion equations for all birds or all mammals.

[f] Best professional judgement based on Lent (1966), Hemming (1987, 1988, 1989, 1991), Pollard (1994a,b).

Table 5-6. Parameters proposed for use with probabilistic exposure analysis for moose

	Distribution	Value
Receptor Population Parameters		
Body mass (kg)	Uniform	Range 340–400[a]
Food ingestion rate (kg dry weight/day)	Functional	Mean 2.73; SD 0.082[b]
Proportion of diet represented by shrubs (percent)	Uniform	90–100[c]
Proportion of diet represented by herbaceous plants (percent)	Functional	100% - % shrubs[c]
Soil ingestion rate (percent of food ingestion rate)	Uniform	Min 0 %; Max 2.0%[d]
Water ingestion rate (L/day)	Functional	Mean 20.3; SD 0.85[e]
Regional Habitat Utilization Parameters		
Home range ($miles^2$)		Range 40–300[f]
Temporal Migration Parameters		
Time use (days)	Absolute value	365[g]

Note: SD - standard deviation

[a] Based on mean values for females reported by Franzmann et al. (1978).

[b] Based on Nagy et al. (1999) allometric equation for herbivores.

[c] Estimated from diets reported for Alaska in Franzmann and Schwartz (1997).

[d] Based on 2 percent soil in moose diet from Beyer et al. (1994).

[e] Based on U.S. EPA (1993) drinking water ingestion equations for all mammals.

[f] Based on home ranges of nonmigratory females in southcentral Alaska (Ballard et al. 1991, as cited in Franzmann and Schwartz 1997).

[g] Assumes receptor is present year-round at the site.

Table 5-7. Parameters proposed for use with probabilistic exposure analysis for Lapland longspur

	Distribution	Value
Receptor Population Parameters		
Body mass (g)	Triangular	Mean 25; Range 22–31[a]
Food ingestion rate (g dry weight/day)	Functional	Mean 6.83, SD 0.34[b]
Proportion of diet represented by terrestrial invertebrates (percent)	Uniform	Min 70% Max 90%[c]
Proportion of diet represented by terrestrial herbaceous vegetation (percent)	Functional	100% - % invertebrates[c]
Soil ingestion rate (percent of food ingestion rate)	Uniform	Min 0 %; Max 1.4%[d]
Water ingestion rate (mL/day)	Functional	Mean 5.1, SD 0.25[e]
Regional Habitat Utilization Parameters		
Home range (ha)	Uniform	1.8–5.5[f]
Temporal Migration Parameters		
Time use (days)	Uniform	108–150[g]

Note: SD - standard deviation

[a] Mean and range of female body weights from Irving (1960).

[b] Based on Nagy et al. (1999) allometric equation for Passerines.

[c] Estimated from summer diets near Barrow, AK (Custer and Pitelka 1978).

[d] Based on 1.4 percent soil in Lapland longspur diet reported by URS Team (1996).

[e] Based on U.S. EPA (1993) drinking water ingestion equations for all birds.

[f] Minimum value based on mean territory of breeding males near Barrow, AK (Seastedt and MacLean 1979); maximum value based on maximum area of 95 percent foraging space for females near Barrow, AK (estimated from Figure 3 in Tryon and MacLean 1980).

[g] Estimated from first and last sightings at Point Barrow, AK (Gabrielson and Lincoln 1959) and at Cape Thompson, AK (Williamson et al. 1966).

Table 5-8. Parameters proposed for use with probabilistic exposure analysis for tundra shrew

	Distribution	Value
Receptor Population Parameters		
Body mass (g)	Triangular	Mean 6.3; range 4.4–12[a]
Food ingestion rate (g dry weight/day)	Functional	Mean 1.28; SD 0.17[b]
Proportion of diet represented by terrestrial invertebrates (percent)	Absolute value	100[c]
Soil ingestion rate (percent of food ingestion rate)	Uniform	Min 2 %, Max 5%[d]
Water ingestion rate (mL/day)	Functional	Mean 1.19; SD 0.23[e]
Regional Habitat Utilization Parameters		
Home range (m^2)	Normal	Mean 2,400; SD 1,209[f]
Temporal Migration Parameters		
Time use (days)	Absolute value	365[g]

Note: SD - standard deviation

[a] Mean and range of vole weights in the Mackenzie Delta Region, NWT, Canada (Martell and Pearson 1978).

[b] Estimated from Buckner (1964) assuming a moisture content of 75 percent in diet from U.S. EPA (1993).

[c] Based on Yudin (1962) not seen, cited in Aitchison (1987), and Buckner (1964).

[d] Best professional judgement based on Beyer et al. (1994).

[e] Based on U.S. EPA (1993) drinking water ingestion equations for all mammals.

[f] Value calculated from data in Hawes (1977) for *Sorex vagrans* and *Sorex obscurus* breeding individuals.

[g] Assumes receptor is present year-round at the site.

Table 5-9. Parameters proposed for use with probabilistic exposure analysis for snowy owl

	Distribution	Value
Receptor Population Parameters		
Body mass (kg)	Normal	Mean 2.279; SD 0.057[a]
Food ingestion rate (kg dry weight/day)	Functional	Mean 0.125; SD 0.0024[b]
Proportion of diet represented by small mammals (percent)	Absolute value	100[c]
Soil ingestion rate (percent of food ingestion rate)	Uniform	Min 0%, Max 2%[d]
Water ingestion rate (L/day)	Functional	Mean 0.10; SD 0.002[e]
Regional Habitat Utilization Parameters		
Home range ($miles^2$)	Uniform	0.5–4[f]
Temporal Migration Parameters		
Time use (days)	Absolute value	365[g]

Note: SD - standard deviation

[a] Mean female body wight from Kerlinger and Lein (1988).

[b] Estimated from Gessaman (1972) and Pitelka et al. (1955) assuming a moisture content of 68 percent in diet from U.S. EPA (1993).

[c] Simplifed from Parmelee (1992).

[d] Based on minimum soil ingestion rate from Beyer et al. (1994).

[e] Based on U.S. EPA (1993) drinking water ingestion equations for all birds or all mammals.

[f] Based on nesting territories near Barrow, AK (Pitelka et al. 1955) and breeding territories on Baffin Island (Watson 1957).

[g] Assumes receptor is present year-round at the site.

Table 5-10. Parameters proposed for use with probabilistic exposure analysis for arctic fox

	Distribution	Value
Receptor Population Parameters		
Body mass (kg)	Normal	Mean 3.2; SD 0.45[a]
Food ingestion rate (kg dry weight/day)	Functional	Mean 0.111; SD 0.013[b]
Proportion of diet represented by small mammals (percent)	Absolute value	100[c]
Soil ingestion rate (percent of food ingestion rate)	Uniform	Min 0%, Max 2.8%[d]
Water ingestion rate (L/day)	Functional	Mean 0.28; SD 0.034[e]
Regional Habitat Utilization Parameters		
Home range (km^2)	Uniform	4.5–48[f]
Temporal Migration Parameters		
Time use (days)	Absolute value	365[g]

Note: SD - standard deviation

[a] Mean female body weight from Anthony (1997).

[b] Based on Nagy et al. (1999) allometric equation for Carnivora.

[c] Simplified from Anthony et al. (2000).

[d] Based on 2.8 percent soil in red fox diet from Beyer et al. (1994).

[e] Based on U.S. EPA (1993) drinking water ingestion equations for all birds or all mammals.

[f] Minimum value is mean home range size for females in western Alaska (Prestrud 1992); maximum value is mean home range size for breeding females in Svalbard, Norway (Prestrud 1992).

[g] Assumes receptor is present year-round at the site.

Table 7-1. Data needs for the ecological risk assessment

Environment	Assessment Endpoint	Representative Receptor	Food Item	Data Need
Tundra	Structure and function of terrestrial plant communities	Terrestrial plant communities	NA	Tundra plant community surveys
Tundra	Structure and function of tundra soil fauna communities	Tundra soil fauna communities	NA	None. Not directly assessed; evaluated through terrestrial plant community analysis
Tundra	Survival, growth, and reproduction of terrestrial avian herbivore populations	Willow ptarmigan	Terrestrial plants	CoPCs in terrestrial plants[a]
Tundra	Survival, growth, and reproduction of terrestrial mammalian herbivore populations	Tundra vole; caribou; moose	Terrestrial plants	CoPCs in terrestrial plants[a]
Tundra	Survival, growth, and reproduction of terrestrial avian invertivore populations	Lapland longspur	Terrestrial invertebrates	CoPCs in terrestrial invertebrates[a]
Tundra	Survival, growth, and reproduction of terrestrial mammalian invertivore populations	Tundra shrew	Terrestrial invertebrates	CoPCs in terrestrial invertebrates[a]
Tundra	Survival, growth, and reproduction of terrestrial avian carnivore populations	Snowy owl	Small mammals	CoPCs in small mammals[a]
Tundra	Survival, growth, and reproduction of terrestrial mammalian carnivore populations	Arctic fox	Small mammals	CoPCs in small mammals[a]
Streams	Structure and function of stream aquatic and wetland plant communities	Stream aquatic and wetland plant communities	NA	Stream aquatic and wetland plant community surveys
Streams	Structure and function of stream aquatic invertebrate communities	Stream aquatic invertebrate communities	NA	Stream aquatic invertebrate community surveys
Streams	Survival, growth, and reproduction of stream avian herbivore populations	Green-winged teal	Aquatic/wetland plants	CoPCs in stream aquatic/wetland plants[a]
Streams	Survival, growth, and reproduction of stream mammalian herbivore populations	Muskrat	Aquatic/wetland plants	CoPCs in stream aquatic/wetland plants[a]
Streams	Survival, growth, and reproduction of stream avian invertivore populations	Common snipe	Aquatic invertebrates	CoPCs in stream invertebrates (lead)
Tundra ponds	Structure and function of tundra pond aquatic and wetland plant communities	Tundra pond aquatic and wetland plant communities	NA	Tundra pond aquatic and wetland plant community surveys

Table 7-1. (cont.)

Environment	Assessment Endpoint	Representative Receptor	Food Item	Data Need
Tundra ponds	Structure and function of tundra pond aquatic invertebrate communities	Tundra pond aquatic invertebrate communities	NA	Tundra pond aquatic invertebrate community surveys
Tundra ponds	Survival, growth, and reproduction of tundra pond avian herbivore populations	Green-winged teal	Aquatic/wetland plants	CoPCs in tundra pond aquatic/wetland plants[a]
Tundra ponds	Survival, growth, and reproduction of tundra pond mammalian herbivore populations	Muskrat	Aquatic/wetland plants	CoPCs in tundra pond aquatic/wetland plants[a]
Tundra ponds	Survival, growth, and reproduction of tundra pond avian invertivore populations	Common snipe	Aquatic invertebrates	CoPCs in tundra pond aquatic invertebrates (cadmium, lead, mercury, thallium, and zinc)
Coastal lagoons	Structure and function of coastal lagoon aquatic and wetland plant communities	Coastal lagoon aquatic and wetland plant communities	NA	Coastal lagoon aquatic and wetland plant community surveys
Coastal lagoons	Structure and function of coastal lagoon aquatic invertebrate communities	Coastal lagoon aquatic invertebrate communities	NA	Coastal lagoon aquatic invertebrate community surveys
Coastal lagoons	Survival, growth, and reproduction of coastal lagoon avian herbivore populations	Brant	Aquatic/wetland plants	CoPCs in coastal lagoon aquatic/wetland plants (lead and zinc)
Coastal lagoons	Survival, growth, and reproduction of coastal lagoon avian invertivore populations	Black-bellied plover	Aquatic invertebrates	CoPCs in coastal lagoon aquatic invertebrates (lead and zinc)

Note: CoPC - chemical of potential concern
NA - not applicable

[a] CoPCs for terrestrial avian and mammalian herbivores, invertivores, and carnivores, and for freshwater aquatic avian and mammalian herbivores are aluminum, antimony, arsenic, barium, cadmium, chromium, cobalt, lead, mercury, molybdenum, selenium, thallium, vanadium, and zinc.

Photographs

Photographs of
Environments in the
Ecological Risk Assessment Study Area

Photograph 1.
Anxiety Ridge Creek

Photograph 2.
Aufeis Creek

Photographs of
Environments in the
Ecological Risk Assessment Study Area

Photograph 3.
Port Lagoon North

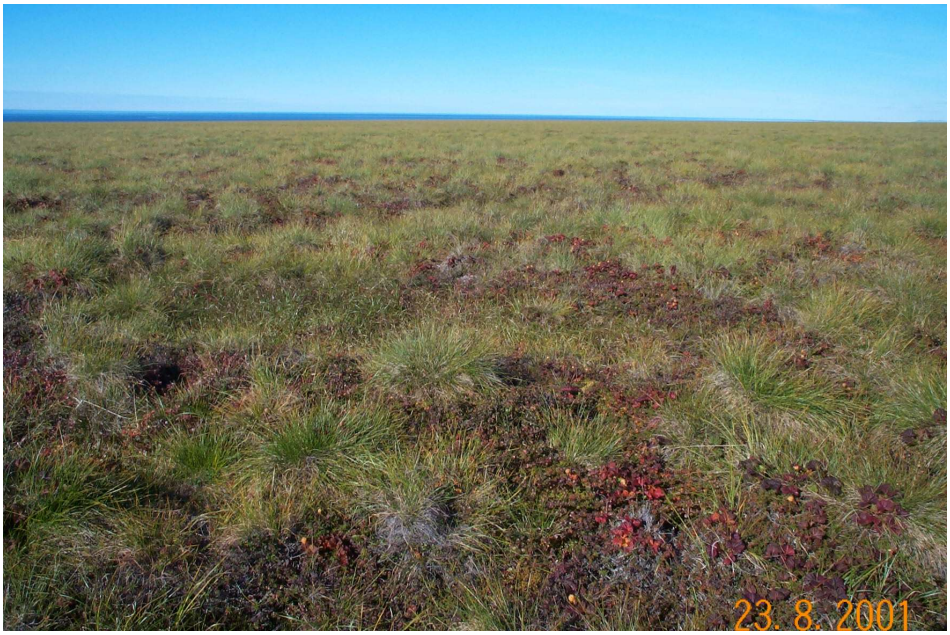
Photograph 4.
Small tundra pond near port facilities

Photographs of
Environments in the
Ecological Risk Assessment Study Area

Photograph 5.
Tundra pond

Photograph 6.
Typical vegetation along the DMTS road

Photographs of
Environments in the
Ecological Risk Assessment Study Area



Photograph 7.
Tussock tundra near the port facility