DMTS Fugitive Dust Risk Assessment Volume I—Report

Prepared for

Teck Cominco Alaska Incorporated
Anchorage, Alaska

Exponent®

DMTS Fugitive Dust Risk Assessment Volume I—Report

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

November 2007

Doc. no.   8601997.007 5400 1107 SS15

Contents

Page

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

ADPH	Alaska Division of Public Health
AIDEA	Alaska Industrial Development and Export Authority
ALM	adult lead model
ANOVA	analysis of variance
AWQC	ambient water quality criteria
BCF	bioconcentration factor
$BMDL_{05}$	benchmark dose associated with the 5 percent response rate
BSAF	biota-sediment accumulation factor
CAKR	Cape Krusenstern National Monument
CCC	criteria 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
CSF	cancer slope factor
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
DRO	diesel-range organic
EPA	U.S. Environmental Protection Agency
EPC	exposure point concentration
EPT	Ephemeroptera, Plecoptera, and Trichoptera
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
NAAQS	National Ambient Air Quality Standards
NANA	NANA Regional Corporation
NMDS	nonmetric multidimensional scaling
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
PCA	principal component analysis
PEC	probable effect concentration
PRG	preliminary remediation goal
RBC	risk-based concentration
RDA	recommended daily allowance
RfD	reference dose
RME	reasonable maximum exposure
RRO	residual-range organic
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
WACH	Western Arctic Caribou Herd
WDOE	Washington State Department of Ecology

Executive Summary

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 exposure to metals in soil, water, sediments, and biota in areas surrounding the DMTS, and in areas surrounding the Red Dog Mine ambient air/solid waste permit boundary. The risk assessment is part of the overall process in which areas of fugitive dust deposition surrounding the DMTS are being evaluated (see the main text Section 1, *Introduction*, for a review of regulatory context). The results of the risk assessment provide a snapshot of risk under current conditions that will help risk managers to determine what additional actions may be necessary to reduce those risks now and in the future.

What This Document Includes

This document presents a revised risk assessment for the DMTS and the area outside of the Red Dog Mine ambient air boundary. The major parts of the risk assessment document include the preliminary human health and ecological conceptual site models, which are presented and then refined based on the results of screening and selection of chemicals of potential concern (CoPCs). Human health and ecological risk calculations are then presented, the risk assessment results are summarized, and a brief discussion of risk management follows. Appendices to the document describe the Phase I and Phase II field programs conducted to provide data for the risk assessment, present data used in the assessment, as well as food-web model and results tables, and also include a chronology of dust control improvements to the DMTS and port operations.

Document History and Public Involvement

This section provides an overview of the history of the risk assessment document, from the development of the conceptual site model, to the draft and final work plan, to the draft and final risk assessment documents.

Conceptual Site Model—A preliminary conceptual site model was included in the Fugitive Dust Background Document (DEC et al. 2002). DEC et al. (2002) also incorporated an appendix documenting specific comments and concerns voiced by village residents in the area of Red Dog Mine. An overview of the conceptual site model was presented to Kivalina and Noatak residents in June 2002. A revised conceptual site model was submitted to DEC in January 2003.

Draft and Final Risk Assessment Work Plan—A draft work plan was submitted to DEC in January 2003 (Exponent 2003b). Following submittal of the draft work plan, a public comment period was held in February 2003, and presentations were made to Kivalina and Noatak residents about the work plan. The revised work plan submitted in February 2004 incorporated revisions based on written and verbal comments and feedback (DEC 2003b) obtained during the public comment period from individuals (e.g., village residents), non-governmental organizations (e.g., Trustees for Alaska, NANA Regional Corporation [NANA]), and government agencies (e.g., DEC, Alaska Industrial Development and Export Authority, National Park Service [NPS]) on the January 2003 work plan. DEC provided comments on the February 2004 work plan in April 2004 (DEC 2004a), and the work plan was approved with response to comments in October 2004 (Exponent 2004b; DEC 2004b).

Draft and Final Risk Assessment—The draft DMTS risk assessment was issued to the DEC in April 2005 (Exponent 2005a). The draft document expanded upon the work presented in the risk assessment work plan (Exponent 2004b), using the framework established in that document, and incorporating revisions agreed to in the response to comments on the work plan (Exponent 2004b; DEC 2004b). After the draft risk assessment was issued to DEC in April 2005 (Exponent 2005a), a public comment period of 45 days followed. Upon closure of the public comment period for the draft DMTS risk assessment, comments had been received from DEC, U.S. Environmental Protection Agency (EPA), NPS, U.S. Geological Survey, NANA, Center for Science in Public Participation, and Alaska Community Action on Toxics. Comment response documents accompany this final risk assessment, and this document incorporates revisions based on the comment responses and comment resolution process conducted by DEC, as lead agency on the risk assessment.

Human Health Risk Assessment Results

A site-specific human health risk assessment (Section 5) was conducted to evaluate exposure to DMTS-related metals through incidental soil ingestion, water ingestion, and subsistence food consumption under three scenarios: 1) child subsistence use, 2) adult subsistence use, and 3) combined worker/subsistence use. The estimated risks from each of the scenarios were within acceptable limits and are summarized below. Risks are necessarily expressed separately for lead and for the other (non-lead) metals because a different methodology is used to estimate lead exposure and risks, as described in Section 5.2.2.1.

Child Subsistence Use

- Using EPA's integrated exposure uptake/biokinetic child lead model (U.S. EPA 1999b), with the model default soil lead bioavailability of 30 percent, the model predicted a geometric mean blood lead level of 1.2 μg/dL, with a less than 0.0005 percent chance of exceeding the target blood lead level of 10 μg/dL.

- Using the site-specific soil lead bioavailability of 9.7 percent, the model predicted a geometric mean blood lead level of 1.0 μg/dL, with a less than 0.0005 percent chance of exceeding the target blood lead level of 10 μg/dL.
- The cumulative hazard index from non-lead CoPCs was 0.3, well below the target hazard index of 1.0.
- Assuming a fractional intake from the site as high as 0.33 (which is 3.7 times the site fractional intake of 0.09), cumulative risks from non-lead CoPCs would not exceed the target hazard index of 1.0.
- The highest hazard index was 0.1 for cadmium exposure from caribou consumption. Assuming a fractional intake from the site as high as 0.95, caribou cadmium related risks would not exceed the target hazard index of 1.0.
- Assuming 100-percent intake from the site (fractional intake=1.0), no other single CoPC would have a risk exceeding the target hazard index of 1.0.

Adult Subsistence Use

- For subsistence use, lead risks were evaluated only for children, but this would also be protective of adult exposure (see results for lead summarized above for child subsistence use).
- The cumulative hazard index from non-lead CoPCs was 0.1, well below the target hazard index of 1.0.
- Assuming a fractional intake from the site as high as 0.93, cumulative risks from non-lead CoPCs would not exceed the target hazard index of 1.0.
- Assuming 100-percent intake from the site (fractional intake=1.0), no single CoPC would have a risk exceeding the target hazard index of 1.0.

Worker/Subsistence Use

- Using the adult lead model default soil lead bioavailability of 12 percent, the model predicted a geometric mean blood lead level in the fetuses of pregnant women of 1.9 μg/dL, with a 1.3 percent chance of exceeding the target blood lead level of 10 μg/dL.
- Using the site-specific soil lead bioavailability of 3.9 percent, the model predicted a geometric mean blood lead level in the fetuses of pregnant women of 1.6 μg/dL, with a 0.7 percent chance of exceeding the target blood lead level of 10 μg/dL.
- The cumulative hazard index from non-lead CoPCs was 0.08, well below the target hazard index of 1.0.

- Assuming 100-percent intake from the site (fractional intake=1.0), cumulative risk from non-lead CoPCs would not exceed the target hazard index of 1.0.

Overall, risks were well within acceptable public health limits. The results of the risk assessment, along with the results from the subsistence foods evaluations (Appendix H), suggest that risks associated with continued harvesting of subsistence foods from the site, including in unrestricted areas near the DMTS, are not significantly elevated. In addition, although harvesting remains off limits within the DMTS, human health risks were not elevated even when data from restricted areas were included in the risk estimates.

Ecological Risk Assessment Results

A site-specific ecological risk assessment (Section 6) was conducted to evaluate risk to ecological receptors inhabiting terrestrial, freshwater stream and pond, coastal lagoon, and marine environments from exposure to DMTS-related metals. The risk conclusions for each habitat are summarized in the following sections.

Terrestrial Environments

- Changes in vegetation community structure are observable within 100 m of the DMTS road and port facilities. These community shifts appear to be, in part, a result of physical and chemical influences of the road and their effect on hydrology, soil chemistry, and plant vitality. Physical and chemical stresses are commonly found associated with gravel roads in tundra environments. The importance of CoPCs in fugitive dust relative to physical stresses caused by the DMTS road in producing these changes could not be determined based on the data available at this time. However, physical factors are likely to exert their greatest influence near the road and facility areas where dust deposition is greatest and drainage may be locally altered, whereas chemical factors (e.g., elevated metals and pH) are likely to become relatively more important at greater distances from dust sources, but may also be significant near the road and port facility areas.

- Differences between reference plant communities and plant communities beyond 100 m from the DMTS road, specifically the 2- to 4.5-fold decrease in lichen cover at 1,000 to 2,000 m from the road, appear to be a result of fugitive dust deposition. Further study would be required to define the full nature and extent of lichen effects related to fugitive dust deposition from the DMTS port, road, and Red Dog Mine, and to identify the causative agent(s) of lichen decline.

- In port facility areas, particularly in the area immediately downwind of Concentrate Storage Building 1 (CSB1), the presence of stressed and dead vegetation appears to be primarily related to fugitive concentrate dust deposition.

- Herbivorous and insectivorous small mammals (e.g., voles and shrews) inhabiting tundra within 10–100 m of the DMTS road, near the port facilities, or near the mine's ambient air/solid waste boundary showed incremental risk from exposure to aluminum and barium. However, exposures decreased to no-effects levels or were comparable to reference exposures beyond 100 m from the road and 1,000 m from the mine's ambient air/solid waste boundary. Although elevated risks were predicted for aluminum and barium near the road, port, and mine, the actual potential for adverse effects is thought to be small given the highly conservative nature of the aluminum and barium toxicity reference values (TRVs) and low bioavailability of aluminum and barium at the site (Shock et al. 2007).

- Adverse effects to herbivorous birds (e.g., ptarmigan) from lead are possible near the port and mine. These effects, if occurring, could result in population-level effects in these areas. However, along the length of the road, the likelihood of adverse effects to herbivorous birds is low.

- For caribou, no adverse effects are predicted for the vast majority (>99.98 percent) of caribou that pass through the site only during migration. Caribou over-wintering near the mine have an estimated exposure to aluminum and barium that is 1.3 to 2.5 times the lowest-observed-adverse-effect levels. However, the actual potential for adverse effects to over-wintering caribou is thought to be small, given the highly conservative nature of the aluminum and barium TRVs and low bioavailability of aluminum and barium at the site (Shock et al. 2007).

- Population-level effects are considered unlikely for other terrestrial wildlife, including large-bodied mammalian herbivores (e.g., moose), avian invertivores (e.g., Lapland longspur and common snipe), and avian and mammalian carnivores (e.g., snowy owl and arctic fox), under current conditions.

Freshwater Stream Environments

- Benthic macroinvertebrate drift assemblages indicated that the overall characteristics of the communities found in the site streams crossing the road were similar to those in reference streams.

- Fish monitoring studies have found no evidence of a road-related effect on metals concentrations in tissue of fish upstream and downstream of the DMTS in the Omikviorok River and Aufeis Creek. However, in Anxiety Ridge Creek near the mine, cadmium and lead concentrations in tissue of juvenile Dolly Varden were significantly higher in fish downstream from the haul road compared with upstream fish, and although the most conservative screening benchmarks for fish tissue were exceeded, concentrations were also within the range of no-effects values from the literature. Thus, adverse effects to fish populations are not predicted in the Omikviorok River and Aufeis Creek, but cannot be ruled out in Anxiety Ridge Creek.

- Metals concentrations in riparian area plants were generally within the range of reference concentrations and/or literature phytotoxicity thresholds. No indications of phytotoxicity were observed in plants at site streams, and plant health appeared similar at site and reference streams.

- The likelihood of adverse population-level effects to wildlife foraging in streams, including avian and mammalian herbivores (e.g., green-winged teal, muskrat, and moose) and avian invertivores (e.g., common snipe), is considered to be very low.

Freshwater Pond Environments

- Adverse effects are not predicted in tundra ponds along the DMTS road, or at distances greater than 100 m from facilities. For these ponds, CoPC concentrations in sediment are not expected to be toxic to benthic macrofauna based on toxicity test data for coastal lagoons. Metals concentrations in plants were generally within the range of reference concentrations and/or below phytotoxicity thresholds, and food-web models indicate a very low likelihood of adverse population-level effects to herbivorous wildlife (e.g., green-winged teal and muskrat) and avian invertivores (e.g., common snipe).

- There is a potential for adverse effects to invertebrates and plants in ephemeral ponds located within 100 m of the concentrate conveyor and other port facilities, although no effects were observed during field sampling in those ponds.

Coastal Lagoon Environments

- Sediment toxicity tests indicated no effects to benthic invertebrates in lagoons, even when exposed to elevated CoPC concentrations in sediments from locations nearest to port facilities.

- Plant community structure was similar at site and reference lagoons. Natural variability among and within lagoon plant communities likely accounts for the few differences that were observed. However, only fringing wetland vegetation was assessed for coastal lagoons, while plant communities with abundant lichen cover were assessed in the terrestrial coastal plain transects.

- The likelihood of adverse population-level effects to wildlife foraging in coastal lagoons, including herbivorous and invertivorous birds (e.g., brant and black-bellied plover), and mammalian herbivores (e.g., muskrat and moose), is considered to be very low.

- No fish were present in port site lagoons, as the lagoons have no open water connections to the Chukchi Sea, and they also freeze solid in the winter.

Marine Environment

- No effects were predicted for receptors in the marine environment because the metals concentrations in sediment and water were below effects levels.

Where We Are in the Process, and What Comes Next

Upon submittal of this revised risk assessment to DEC, the agency will issue a decision regarding acceptance of the risk assessment. Following completion of the risk assessment, a risk management plan will be developed to address the issues identified by this risk assessment, which are summarized above. The plan will include evaluation of risk management options within the general categories of institutional controls, engineering controls, monitoring, and remediation/restoration. The plan will identify the most appropriate combination of actions to achieve the overall goal of minimizing risk to human health and the environment surrounding the DMTS and outside the Red Dog Mine boundary over the life of the mine.[1] Development of the plan is anticipated to be a collaborative process involving DEC and other stakeholders throughout the process of identifying, defining, and refining objectives, and evaluating options and methods to achieve those objectives.

[1] Note that the mine closure and reclamation plan will address risk management within the mine boundary.

1 Introduction

Elevated metals concentrations have been identified in tundra in areas surrounding the DeLong Mountain Regional Transportation System (DMTS[2]) and Red Dog Mine, primarily as a result of fugitive dust[3] deposition. The purpose of the DMTS fugitive dust risk assessment is to estimate the magnitude and likelihood of unacceptable risks to human and ecological receptors posed by exposure to metals in soil, water, sediments, and biota in areas surrounding the DMTS, and in areas surrounding the Red Dog Mine ambient air/solid waste permit boundary.

The risk assessment was 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 provide a snapshot of risk under current conditions that will help risk managers to determine what additional actions may be necessary to reduce those risks now and in the future. In the interim (while the risk assessment is being completed), a number of actions have been and are being taken by Teck Cominco Alaska Incorporated (Teck Cominco) to reduce 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 (reflecting the mineralization) 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). Topography and water features are illustrated in Figure 1-6. 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 ambient air/solid waste permit boundary around the mine.[4] The approximate area of focus in the risk assessment is highlighted in Figure 1-7.

---

[2] 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.

[3] "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. Near the mine, fugitive dust may originate from various sources within the mine, including blasting in the pit, ore stockpiles, waste rock dumps, tailings pond sediments (historically), and road dust from truck traffic, which may also include some ore concentrate dust.

[4] The mine area within the permit boundary (shown in Figure 1-5) is not addressed in this 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 (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 1-8). The barges have built-in and enclosed conveyors that are used to transfer the concentrates to the holds of deepwater ships.

Moss studies performed in 2000 and 2001 by the National Park Service (NPS) (Ford and Hasselbach 2001; Hasselbach 2003, pers. comm.; Hasselbach et al. 2005) 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. A fugitive dust background document was published in spring 2002, providing an overview of local observations and concerns, local and regional background information, Red Dog operations, regulatory history, environmental data, nature and extent of fugitive dust, a preliminary conceptual site model (CSM) for the risk assessment, and review of regulatory and decision-making frameworks for addressing the fugitive dust issue (DEC et al. 2002).

Teck Cominco completed additional characterization at the port site in 2002 (Exponent 2003c; Teck Cominco 2003a). Sampling programs designed to support the risk assessment were conducted in 2003 and 2004 to obtain data for additional analytes in multiple environments and media. These programs are described in the field sampling plans (Exponent 2003e, 2004a), and in appendices to this document.

The nature and extent of dust deposition has been evaluated in these prior studies by Exponent and NPS, as listed above. Some key observations are summarized here:

- Moss data collected during various sampling efforts by NPS and Teck Cominco, when presented together (Figure 1-9), effectively illustrate the primary source areas and deposition patterns in the vicinity of the DMTS corridor and mine. The moss concentration patterns illustrate how the prevailing wind patterns originating from the southeast to northeast result in greatest deposition to the north and west of DMTS and mine facility areas.

- Within the DMTS facility areas, metals concentrations decrease away from facility sources (Figure 1-9), and vary along the length of the road corridor, with the highest concentrations near the port and the mine, as a result of concentrate tracking that has historically occurred with haul trucks exiting the CSBs at the mine and port (Figure 1-10).

Many improvements have been made over the years by Teck Cominco to reduce fugitive dust emissions. Broadly, these include improvement to engineering controls and enclosures around ore crushing, milling, concentrate storage and loading at the mine, as well as concentrate trucking and storage, conveyance, bargeloading, and shiploading facilities at the port. In addition to physical dust control improvements, procedural improvements have been made as well. Further description of these measures, as they pertain to the risk assessment CSM, is provided in Section 2.2.4. Teck Cominco continues to work on additional dust control improvements on an ongoing basis.

1.2 Document Organization

The sections of the risk assessment include:

- Section 1, *Introduction*
- Section 2, *Preliminary Conceptual Site Model*
- Section 3, *Selection of Chemicals of Potential Concern*
- Section 4, *Supplemental Data Collection for Risk Assessment*
- Section 5, *Human Health Risk Assessment*
- Section 6, *Ecological Risk Assessment*
- Section 7, *Calculation of Risk-Based Cleanup Levels*
- Section 8, *Conclusions*
- Section 9, *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 Used in CoPC Screening*
- Appendix D, *Organic Chemical Data*
- Appendix E, *Summary of Phase II Sampling Program for the DMTS Fugitive Dust Risk Assessment*
- Appendix F, *Data Quality Review for the Phase II Sampling Program for the DMTS Fugitive Dust Risk Assessment*
- Appendix G, *Additional Data Used in the Risk Assessment*
- Appendix H, *Subsistence Foods Data Evaluations*

- Appendix I, *Vegetation Community Surveys*
- Appendix J, *Photographs of Typical Biota Samples*
- Appendix K, *Food-Web Model Tables*
- Appendix L, *Chronology of Dust Control Improvements to the DMTS Road and Port Operation.*

2 Preliminary Conceptual Site Model

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 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 Mine 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 (Exponent 2003a,b).

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 sources of chemicals of interest are the ore concentrates 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. The zinc and lead concentrates produced at the Red Dog Mine are sulfides, and they include minor amounts of other metal sulfides and impurities. Typical concentrations of constituents in the concentrates are illustrated in Table 2-1. The zinc concentrate contains approximately 55 percent (550,000 ppm) zinc, 0.33 percent (3,300 ppm) cadmium, which is associated with the zinc mineral, and 3.2 percent (32,000 ppm) lead. The lead concentrate contains approximately 58 percent (580,000 ppm) lead, 0.12 percent (1,200 ppm) cadmium, and 10.8 percent (108,000 ppm) zinc. The concentrates are a very fine powder. Particles are smaller than 40 μm in size, with more than 80 percent smaller than 20 μm in size (Teck Cominco 2003b,f).

The mineral composition of the concentrates is as follows (DEC et al. 2002):

- Zinc Concentrate—80 to 85 percent sphalerite, 7 to 9.5 percent pyrite, 2.5 to 5 percent galena, and 2.8 to 3.7 percent quartz

- Lead Concentrate—60 to 70 percent galena, 14 to 21 percent sphalerite, 6 to 15 percent pyrite, and 2 to 4.5 percent quartz.

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. This table includes only spills from DEC's database that have occurred within the DMTS road and port areas that are subject to this risk assessment (i.e., excluding those within the mine boundary). 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 (see Figure 1-6). 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-hand column of Table 2-2). 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 pavement provides a barrier preventing exposure to any residual petroleum hydrocarbons in soil. The concrete apron and collection sump are part of a spill containment system that is maintained on an ongoing basis by Teck Cominco as part of their spill prevention program.

The large diesel spill (originally estimated at 36,000 gallons, later estimated at approximately 22,000 gallons) occurred from Fuel Storage Tank #2 (Tank 2) at the port site (see Figure 1-6). Although that spill is recorded as cleaned up, it was unclear what the final concentrations were at the time that DEC issued a "No Further Action" letter. Therefore, samples were collected at the former Tank 2 spill area as part of the 2003 field sampling program (Exponent 2003e and Appendix A of this document). The samples collected in the Tank 2 spill area were collected from a localized tundra area adjacent to the Tank 2 containment. Samples were collected at three depth intervals, the first of which is the first 0 to 2 cm beneath the live vegetation mat. The second was collected between 2 cm and the bottom of the organic tundra soils, and the third was collected from inorganic substrate soils below the organic tundra soils (if present), or from just above the permafrost. Samples were collected from similar depth intervals in a reference area away from any anthropogenic activity.

Although there were some samples in the Tank 2 area with residual-range organic and diesel-range organic compounds RRO and DRO concentrations elevated above one-tenth Arctic Zone cleanup levels, there are several reasons why this does not warrant retaining DRO and RRO as CoPCs:

1. The former Tank 2 spill area is very localized; it is not a large area.
2. A significant portion of the DRO and RRO concentrations are the result of naturally occurring organic matter. This is illustrated by the results for the reference samples (see data tables in Appendix D). All three of the RRO and two of the three DRO results in the reference samples are above one-tenth of the Arctic Zone cleanup levels as a result of naturally occurring organic matter.
3. The depth intervals in which the elevated values occur are the deeper sample intervals, not the shallow samples. Therefore, there is not a complete exposure pathway for humans or animals that might cross this tundra area.
4. Degradation will continue to reduce residual hydrocarbon concentrations in this area.
5. No activities are planned in this area. However, in the event that any development were to occur in this tundra area, such development would involve placement of additional fill, rather than excavation. Engineering requirements dictate that facilities in this region are constructed on fill above the permafrost. Any utilities would be either within the gravel fill or above grade, because of the presence of permafrost.
6. Beneath the containment area around the tank, any residual hydrocarbons that may remain after historical excavation and treatment activities are at least several feet below the current grade, under clean gravel.

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.). Because of 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 June 7, 1996; Spill No. 97389923301 on August 21, 1997; Spill No. 98389932501 on November 21, 1998; and Spill No. 98389903801 on February 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 August 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 October 5, 1997. Teck Cominco's records show that 500 gallons of process water were spilled on October 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 has been conducted by Teck Cominco (Teck Cominco 2005a) 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 this risk assessment. The characterization process was completed in 2003, and recovery and recycling (where necessary) have been completed (Exponent 2002c,d; 2003d). Results were reported to DEC per the requirements of the Settlement Agreement identified above. Table 2-3 provides summary information about each of the truck spills, and Table 2-4 lists the closeout dates of the re-evaluation of each spill site, and the specific documents containing the closeout information. In general, 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. Dust emanating from concentrate haul trucks or handling facilities may have a higher proportion of fine concentrate particles than dust emanating from road surfaces, which also contains concentrate particles. 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 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 (including replacement of the truck and trailer fleet in fall 2001) have reduced these sources and transport mechanisms, as described in Section 2.2.4.

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 exposed or snow-covered tundra surfaces (e.g., from tundra along the road) where it had previously been deposited.

- **Spray from road traffic—**Under very wet conditions the road becomes saturated and passing vehicles release the finest fraction of the road surface as a mist that is sprayed or otherwise deposited on the adjacent tundra.

- **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 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 as a result of 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—**Runoff 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. 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. In addition to the mechanisms described earlier for the road, which also apply here, the following list includes a number of sources of metals and current and past transport mechanisms associated with mine operations. Dust control measures described in Section 2.2.4 have significantly reduced many of these sources and transport mechanisms:

- **Dust generated by open pit mining activities—**Dust can be generated from drilling, blasting, material handling, and truck haulage activities in the open pit.

- **Dust emissions from materials handling—**Dust can be generated from materials handling activities outside of the open pit, including truck haulage activities, placement of waste rock on waste rock stockpiles, and the stockpiling of ore.

- **Dust emissions from mill and concentrate storage facilities—**Dust can be generated from the ore crushers, the coarse ore stockpile building, and from concentrate storage and loading operations in the CSB. However, significant upgrades have been made to reduce emissions from those facilities over the years, as described in Section 2.2.4.

- **Mechanical or wind-generated dust from surfaces—**Windblown dust can be generated from surfaces around the mine, including the access roads and yards, the tailings beach (historically; see Section 2.2.4), and other mineralized surfaces.

Outside of the mine area, 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 exposed or snow-covered 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 are a result of dust control measures taken with facilities in the mine area, with trucking on the road, and with unloading, storage, transfer, bargeloading, and shiploading facilities at the port. The changes include the use of newer trucks, significant upgrades to the surge bin and truck loading and 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 loading and unloading buildings, and by truck washing in the summer and traffic separation at the mine. Since fall of 2001, concentrate spillage and escapement has been significantly reduced 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 physical and procedural controls implemented to limit tracking, as well as recovery and recycling of metals-containing road material. 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. Efforts to reduce fugitive dust emissions are ongoing. A chronological summary of dust control improvements is provided in Appendix L.

Fugitive dust control improvements have also been made in the mine area. In 1992 a significant number of control measures were implemented. The coarse ore stockpile was enclosed to prevent the escape of fugitive dust, the mine CSB was modified to include a loading bay to reduce tracking, take-up pulleys were relocated to inside the mill or enclosed in place, and a large water truck was purchased to facilitate implementation of additional dust control measures (watering and palliative application) on roads and yards. More recently, a procedural change was made 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. Additional dust controls have also been implemented in the truckloading at the mine CSB, to minimize dust getting on the exterior of the trucks and to reduce tracking from the mine. Traffic separation was implemented in 2004 to separate DMTS road traffic (e.g., concentrate haul trucks) from mine area traffic (e.g., mine vehicles and equipment). A more detailed list of dust control improvements at the mine is provided as an appendix in the recent document *Summary of Mine-Related Fugitive Dust Studies* (Teck Cominco 2005b), and included in Appendix L of this document. Other possible control measures are being evaluated in an ongoing effort for continual reduction of fugitive dust emissions from mine, road, and port facilities.

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, developed at the start of the assessment, 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. The CSM is refined further in Section 5.1 based on the results of screening evaluation and the site-specific knowledge acquired through Phase I and supplemental (Phase II) sampling.

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 land belonging to NANA (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.[5] The DMTS road runs through lands owned by the State of Alaska, NANA, and the federally owned Cape Krusenstern National Monument (CAKR), which is administered by NPS. NANA traded valuable lands it received under the Alaska 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 passed Public Law 99-96 in 1985, which granted a 100-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, 50 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.

Mining at Red Dog is likely to continue approximately another 25 years based solely on current reserve deposit life. However, there are additional deposits that may be viable and continued mining is likely. There are currently no zoning restrictions on land use, and considering the likely continued use as an industrial facility, they are not likely to be necessary. However, zoning restrictions could be considered for certain areas if needed to protect future land users.

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.

---

[5] 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 for the infrastructure.

- **Subsistence hunting and gathering—**Subsistence hunting and gathering is very important to the economic, nutritional, and spiritual well-being of northwest Alaskan residents. As of the early 1980s, approximately one-third of local households were dependent on subsistence hunting and gathering, and 55 percent of these households obtained more than half of their food supply by hunting, fishing, and gathering (U.S. EPA 1984). Although the rate of subsistence use may have declined somewhat since that time, data from the 1990s indicates that subsistence foods continue to provide a substantial portion of the diet, as described in detail in Section 5.2.2.2.3. Subsistence hunting and gathering occurs in the general vicinity of 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.[6]

- **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) is evaluated in the subsistence use scenario. In addition, individuals of all ages are assumed to be able to access soils and subsistence resources along the DMTS. This type of exposure is evaluated as part of the subsistence hunting and gathering land use exposure pathways.

Recreational use of the area is also possible, although recreational use of the DMTS is not permitted. Recreational activities that are usually undertaken in the area of the DMTS 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 1995). The drinking water for the areas under consideration comes from surface water resources.

---

[6] There are public access control plans 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 is assumed for the risk assessment work and for screening steps described here. Further discussion is provided in the human health risk assessment (Section 5) regarding the assumptions about the proportion of subsistence use that occurs near the DMTS.

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.

2.3.2.1.1 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 blood 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 assess a worker scenario for the DMTS transportation corridor, a hypothetical worker is evaluated. This scenario considers exposure to soil and dust based on concentration data for current conditions along the DMTS corridor.

2.3.2.1.2 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 is evaluated in the risk assessment. Workers would not be considered likely to consume as many subsistence foods as individuals who engage in a subsistence lifestyle full-time.

2.3.2.1.3 Workers' Cumulative Exposure

The risk assessment estimates 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 is a hypothetical female worker who comes in contact with lead in site media during pregnancy. The adult lead model (ALM) 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) are 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 are evaluated for non-lead metals, but using standard DEC and U.S. Environmental Protection Agency (EPA) risk assessment methodology, as described in Section 5.2, with input parameters appropriate to an adult worker's potential exposure.

This approach provides an assessment of cumulative worker/subsistence user exposure to CoPCs that are not assessed under the biomonitoring program. In addition, it provides 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 affect 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 the drinking water source in the assessment. Use of other water sources during subsistence activities will be evaluated for the subsistence user scenario, as discussed in Section 2.3.3.2.

Ambient air modeling performed during the air permitting process has demonstrated that air concentrations at and beyond the mine, road, and port ambient air boundaries (see Figure 1-5) do not exceed National Ambient Air Quality Standards (NAAQS).[7] In addition, air monitoring

[7] 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 Safety and Health and/or Mine Safety and Health Administration 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 1-5. The ambient air boundary for the road is located 300 ft on either side of the road centerline.

programs conducted at the mine have demonstrated that concentrations are below the NAAQS both inside and at the ambient air boundary (Teck Cominco 2005b). However, one year of ambient air monitoring for lead was conducted both in Kivalina and in Noatak, partly in response to community concern. Lead data were collected and air concentrations of lead were compared with NAAQS. Quarterly average concentrations were hundreds of times (191 to 387 times) below the NAAQS (Teck Cominco 2004h,i).

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 are quantified in the HHRA. Secondary exposure pathways are those not expected to contribute significantly to risk estimates. Secondary pathways are 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 is further refined following screening for CoPCs.

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:

- Plants: berries, sourdock
- Mammals: caribou
- Birds: ptarmigan
- Freshwater fish: Dolly Varden

- 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 5.2.1.2 of the HHRA.

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 (ADPH 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" (ADPH 2001). Further berry sampling conducted by DEC and Exponent suggested elevated concentrations of some metals at the port site relative to reference conditions near Noatak and Point Hope (Exponent 2002a). More recent data collected during the 2004 field season (Section 4) indicate that berry concentrations may have declined (Appendix H). 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 is 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 (2003a) implicitly acknowledges the relative importance of ingestion and the limited contributions of inhalation and dermal exposure to overall soil and dust metals exposure by calculating cleanup levels only for soil ingestion of metals, and 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 soil cleanup levels specifically protective of dermal or particulate inhalation exposure, with the exception of lead and mercury. However, the soil inhalation cleanup level for mercury appears to be based on volatilization of elemental mercury, which is not present at the site. The form of mercury that would be in site soils is inorganic mercury, which is not volatile and does not have a reference concentration for inhalation exposure, and thus does not have an inhalation cleanup level.[8] The lead cleanup level listed under the inhalation column is the default residential cleanup level for lead derived using EPA's integrated exposure uptake/biokinetic (IEUBK) lead model and is not, strictly speaking, based on inhalation. Rather, it is based on multipathway exposure and is primarily driven by soil ingestion, including the dust that is inhaled and subsequently ingested. Issues related to the relative importance of ingestion of, inhalation of, and dermal contact with soil and dust are described further below.

2.3.3.1.1 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 are 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 thus includes the portion directly ingested, the portion inhaled, and the portion absorbed through the skin.

2.3.3.1.2 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

---

[8] Excluding a specific source of elemental mercury, inorganic mercury is the predominant form of mercury found in soil. U.S. EPA (1997c) states that soil conditions are typically favorable for the formation of inorganic Hg(II) compounds such as HgCl, Hg(OH) and inorganic Hg(II) compounds complexed with organic anions. U.S. EPA (1997c) further notes that 97–99 percent of total soil mercury is in the form of inorganic Hg(II) complexes, with only a small fraction present as elemental mercury in typical soil. Approximately 1–3 percent of the total mercury in typical surface soil is methylmercury, as is the case for Hg(II) species, it will be bound largely to organic matter. In addition, as shown in Table 3-14 of the risk assessment, mercury concentrations in soil at the site were below one-tenth of the DEC soil cleanup level (26 mg/kg), which is based on the inhalation pathway. Thus, mercury would not be considered a CoPC regardless of whether the mercury present is in the elemental or inorganic form.

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 indicated that 98 percent of soil particles are larger than 1 micron in diameter. Approximately 80 percent of soil particles were larger than 10 microns. These particle sizes are from soil samples that were taken along the length of the road from fine material settled at the toe of the road shoulder (Exponent 2002a). Although dust emanating directly from concentrate haul trucks or handling facilities may have a higher proportion of fine concentrate particles than dust emanating from soil and tundra surfaces, the majority of dust exposure would be to a mixture of dust particles resuspended from soil or tundra surfaces. Thus, the majority of inhaled dust and soil at the DMTS would be expected to be ingested or expelled through mucus.

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 2-5 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 *Soil Screening Guidance: 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 2-5, 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), which 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.

2.3.3.1.3 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 *Risk Assessment Guidance for Superfund: Volume 1: Human Health Evaluation Manual (Part E, Supplemental Guidance for Dermal Risk Assessment*) (U.S. EPA 2004), 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 (2004) 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 (2004) 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 (2004), EPA Region 9 calculated PRGs for dermal exposure to arsenic and cadmium. As indicated in Table 2-5, 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 substantial influence on site risks as a result of higher consumption rates relative to soil, the impact of the dermal contact with the soil pathway on the overall assessment is further reduced. Currently, U.S. EPA (2004) provides dermal absorption fractions for only two metals, arsenic and cadmium, and recommends that other inorganic compounds be treated qualitatively.

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 affected 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 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 in comparison with Alaska DEC drinking water maximum contaminant levels and EPA risk-based screening levels (i.e., Region 9 PRGs) (ADPH 2001). Because some streams crossing the DMTS flow into the Wulik River, which in turn provides drinking water for Kivalina, and because some use of drinking water from streams occurs during subsistence use activities, drinking water consumption from the freshwater environment has been identified as a primary exposure pathway for residents and is evaluated in both the subsistence use and the combined worker/subsistence use scenarios. Water data used in the HHRA are from creeks that cross the haul road. These data are expected to reflect surface water that is potentially the most affected by dust or runoff from the DMTS. As a result, use of these data in the assessment is also expected to be protective of subsistence use of other water sources elsewhere in the surrounding area, including water from the Umayutsiak Creek south of the port, where Kivalina residents have indicated some use of water during subsistence activities.

Surface water data are compared with reference conditions and with RBCs protective of residential drinking water. For CoPCs present at concentrations above reference conditions and the RBCs, exposure through ingestion of drinking water is quantified in the risk assessment. Surface water data are also compared to water quality criteria protective of people consuming water and fish, where available. For CoPCs present at concentrations above reference conditions and the water quality criteria, exposure through fish consumption is 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. Nevertheless, seafood consumption from the lagoon and marine environments was initially identified in the preliminary CSM as a primary exposure pathway for subsistence users during preparation of the risk assessment work plan (Exponent 2004b).

2.3.3.3.1 Lagoon Environment

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 because of the low water temperature, which precludes direct contact through swimming and wading, and because there are no fish or shellfish species in these lagoons that are harvested for human consumption. Thus, the lagoon environment does not have complete exposure pathways and will not be included in the quantitative risk assessment. However, to provide a health-protective evaluation of lagoon conditions for the risk management process, metals concentrations in lagoon water near the port are compared with reference conditions, and with water quality criteria protective for human consumption of fish.

2.3.3.3.2 Marine Environment

Little or no direct human contact with marine sediments and water is expected because of the lack of exposed sediment at the site, and the low water temperature, which precludes direct contact through swimming and wading. However, to provide an evaluation of marine conditions that is protective for human consumption of fish and shellfish collected from the marine environment, metals concentrations in marine water and sediment near the port are compared with reference conditions, and with water quality criteria, to determine whether further quantitative risk estimates are necessary 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, developed at the start of the assessment, 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 CSM is refined further in Section 6.1 based on the results of the screening assessment and the site-specific knowledge acquired through Phase I and supplemental (Phase II) sampling. 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 (ERA).

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 1-6). 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 1-6). 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 flow to the north. Streams that cross the northern portion of the road are tributaries to the Wulik River, while streams that cross the southern portion of the road (e.g., Omikviorok River, Straight Creek, Aufeis Creek, and New Heart Creek) flow into Ipiavik Lagoon, north of the port site. The tributaries in this area tend to have high flows in the spring as a result of snowmelt 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 1995). 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 1995; 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).

The study area is habitat for a variety of marine and terrestrial wildlife. Historical documentation such as the 1981−1982 baseline study by Dames and Moore (1983a,b) describes the wildlife observed. A total of 104 species of birds were observed near the mine, port, and DMTS road area during the baseline study. In particular, the sedge-grass habitats found in lakes, ponds, and lagoons along the DMTS road were ideal for Canada geese and other water-oriented birds. Peregrine falcon sightings were documented around the mine area.

Near the port lagoon, birds that were observed included tundra swan (*Cygnus columbianus*), sandhill crane (*Grus Canadensis*), red knot (*Calidris canutus*), semi-palmated sandpiper (*Calidris pusilla*), western sandpiper (*Calidris mauri*), glaucous gull (*Larus hyperboreus*), savannah sparrow (*Passerculus sandwichensis*), raven (*Corvus corax)*, water pipit (*Anthus spinoletta*), yellow wagtail (*Motacilla flava*), oldsquaw (*Clangula hyemalis*), black scoters (*Melanitta nigra*), and dunlins (*Calidris alpinus*) (Dames & Moore 1983a). In the summer 2004 sampling event (Section 4.0 and Appendix E), Canada goose (*Branta Canadensis*), red-necked phalarope (*Phalaropus lobatus*), arctic tern (*Sterna paradisaea*), oldsquaw, Lapland longspur (*Calcarius lapponicus*), loon (*Gavia* sp.), red fox (*Vulpes fulva*), moose, grizzly/brown bear (*Ursus arctos*) (tracks were observed on shore), sandhill cranes, and red breasted mergansers (*Mergus serrator*) were observed in the port area. Historically, the only fish found in the port lagoons was the ninespine stickleback (*Pungitius pungitius*) (Dames & Moore 1983a). The site lagoons are closed to the ocean, and have limited freshwater input from surface water drainages. However, other lagoons in the vicinity that are open to the ocean can be important habitat for anadromous fish (Dames & Moore 1983b).

Caribou (*Rangifer tarandus)*, moose, and muskox (*Ovibos moschatus*) sightings are frequent at the foothills of the DeLong Mountains and along riparian areas of the DMTS road. Moose and muskox are resident in the area. Caribou are primarily migratory, although some individuals may over-winter in the mine area. The grizzly/brown bear has also been seen in various habitats along the DMTS road and mine site. Small mammals, such as the tundra vole (*Microtus oeconomus*), were observed in shrub tundra areas of the DMTS road and evidence of the singing vole (*Microtus miurus*) was found in the mine area (Dames & Moore 1983a,b). Other terrestrial mammals that have historically been observed include the Dall sheep (*Ovis dalli dalli)*, wolf (*Canis* sp.), wolverine (*Gulo gulo*), red fox (*Vulpes vulpes*), arctic ground squirrel (*Spermophilus parryii*), porcupine (*Erethizon dorsatum*), lemming (*Dicrostonyx sp.*), arctic fox (*Alopex lagopus*), lynx (*Lynx* sp.), ermine (*Mustela erminea*), river otter (*Lontra canadensis*), and muskox (*Ovibos moschatus*) (Dames & Moore 1983a,b; RWJ 1997). Marine mammals were observed along the nearshore waters of the port site, including ringed seals (*Phoca hispida*), bearded seals (*Erignathus barbatus*), and spotted seals (*Phoca largha*), of which the ringed seals were most abundant. Dames & Moore (1983a,b) also reported occasional sightings of walruses (*Odobenus Rosmarus*), beluga whales (*Delphinapterus leucas*), harbor porpoises (*Phocoena phocoena*), bowhead whales (*Balaena mysticetus*), and grey whales *(Eschrichtius robustus*) Birds in the open water, such as sea ducks, king eiders (*Somateria spectabilis)*, oldsquaw, and long-tailed jaeger (*Stercorarius longicaudus)*, were also observed (Dames & Moore 1983b).

The most abundant migratory fish species in the nearshore area are chum salmon (*Oncorhynchus keta*) and arctic char (*Salvelinus alpinus*), which use the open lagoons as a transportation corridor between their spawning rivers and the Chukchi Sea. Fish species occurring near the port area include starry flounder (*Platichtys stellatus*), arctic flounder (*Liopsetta glacialis*), saffron cod (*Eleginus gracilis*), Atka mackerel (*Pleurogrammus monopterygius*), rainbow smelt (*Osmerus mordax dentex*), nine-spine stickleback (*Pungitius pungitius*), Pacific herring (*Clupea harengus pallasi*), surf smelt (*Hypomesus pretiosus*), and larval smelt (Family *osmeridae*) (Dames & Moore 1983b).

Marine life such as crabs (helmet crab [*Telmessus cheiragonus*] and red king crab [*Paralithodes camtschaticus*]) and shrimp species (*Caridae* spp., *Crangonidae* spp.) have also been observed in waters offshore of the port. The seastar (*Asterias amurensis*, *Evasterias echinosoma*) is the single most abundant animal at the bottom of the marine waters, followed by the helmet crab. Red king crab are not abundant as a result of the lack of suitable bottom habitats. Marine worms dominate in the sediment (RWJ 2001).

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 an 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 No. 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 for eiders. 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 No. 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 the CAKR, 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 1998a). 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 ERA (Figure 2-2). The CSM only identifies routes by which receptors are exposed to CoPCs and makes no conclusions regarding the potential for risk associated with the exposure pathways.

Primary exposure pathways are those expected to contribute most to total exposure, while secondary exposure pathways are not expected to increase exposure substantially. Primary exposure pathways for terrestrial receptors (such as herbivorous, invertivorous, and piscivorous birds and mammals) include the consumption of plant material or prey and the incidental ingestion of soil or sediment. Secondary exposure pathways for terrestrial receptors include dermal contact with and ingestion of surface water and inhalation of soil particles. 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 (McGeer et al. 2004). In general, inhalation of particles is assumed to be insignificant compared to other exposure routes for metals and is typically not addressed in an ERA (Newman et al. 2003). Therefore dermal contact is not considered a pathway for terrestrial receptors, and inhalation is considered a secondary pathway.

For terrestrial plants, the primary pathways of exposure are contact with and uptake of metals incorporated into soil and uptake of metals deposited onto plant surfaces as fugitive dust (Figure 2-2). Soil fauna are primarily exposed to metals through direct contact with and uptake of the soil and via ingestion of food in the soil.

For aquatic plants, the primary pathways are direct uptake of sediment and surface water, and contact with surface water. Primary exposure pathways for aquatic receptors such as fish and aquatic invertebrates 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). Secondary exposure pathways for aquatic receptors include contact with sediment. Some aquatic receptors may also be exposed through the uptake of metals from sediments.

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 2-6; 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* (Appendix A of DEC 1999).

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 (see Section 3.5.6 for the screening assessment TRVs and Section 6.5.2 for the risk assessment TRVs). Preliminary assessment endpoints, measurement endpoints, and representative receptors are summarized in Table 2-7.

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 principle, which 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.

Exclusion of bismuth, calcium, chloride, gallium, germanium, gold, silicon, sulfate, and sulfur from the risk assessment introduces additional uncertainty into the risk assessment. However, the ability to address that uncertainty is limited because of the lack of adequate toxicological information for these constituents. This lack of toxicological information means that quantitative risk estimates are not possible. On the other hand, the impact of this uncertainty on the overall results of the risk assessment is minimized by the fact that these constituents are generally not considered to be environmental hazards. Thus, while the uncertainty inherent in excluding these constituents is acknowledged, they are not included in the tables used in the remainder of the risk assessment in the interest of clearly and concisely focusing on the CoPCs for which risk can be evaluated. Regarding sulfur, some forms adversely affect non-vascular plants; this issue is discussed in more detail in Section 6.6.3, *Uncertainties Related to CoPC Screening*.

As agreed upon in discussions with DEC, pH was measured in tundra soil samples and at all surface water bodies where sampling was conducted in the 2004 field season (see Section 4). Tundra pH may be influenced by road dust deposition and may contribute to effects on plant communities, thus the need for measuring pH in tundra environments. Recognizing that pH will likely vary naturally in different tundra environments, pH was also measured at reference area stations to provide data for further comparison and evaluation. Section 4 discusses the supplemental data collection for the risk assessment data needs.

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. Further discussion was provided in Section 2.1.2. 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,[9] sediment, and water data that were used in the CoPC screening and that were available for use in the risk assessment at the time the risk assessment work plan was prepared. After the completion of the work plan, a supplemental sampling program was conducted to support the risk assessment. Data used in the CoPC screening are reviewed by environment and medium in the following subsections. Section 4 describes subsequent data collection for additional risk assessment data needs.

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, as well as state and federal agencies, between 1978 and the end of 2003 (supplemental data collection is described in Section 4). 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:

[9] 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.

- **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, in terms of both environmental conditions (e.g., climate and weather) and 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 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.

- **Paving or Removal—**Soil samples that have been removed by excavation (i.e., for recovery and recycling) or that are isolated beneath pavement (Exponent 2002b), were excluded from the screening, because they no longer represent an exposure medium for human or wildlife receptors. Work conducted in 2002 within the port site on the loop road at the truck unloading buildings and approximately the first 6 miles of the DMTS road involved removal and recycling of road surface soil with lead concentrations above the Arctic Zone standard of 1,000 mg/kg, followed by subsequent paving in a pavement test project (Exponent 2002b).

- **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 2003a) 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 (2003a) 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—**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 2003a; Hall 2003, pers. comm.); however, 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 (2003a) 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 are 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 the soil and tundra soil data used in the CoPC screening for the terrestrial environment. Note that "soil" refers to inorganic soil, principally found on the road and facility areas (e.g., gravel roads and pads). "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 used in the CoPC screening included samples from road and facility areas (Figure 3-2). The types of samples on the road included road surface and core samples, and road shoulder samples (fine material from the toe of the road embankment). Surface soil sample results were available for the port facility areas.

3.2.3.2 Reference Soil

The reference inorganic soil samples were 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). The material site samples were composite samples collected from representative source material within each material site from beneath surface layers, where no excavation had previously been done, so there was no exposure to dust deposition.

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). These data were used to select CoPCs for tundra soil, as described below in Section 3.5.1.

3.2.3.4 Reference Tundra Soil

Reference tundra soil samples were collected from the Phase I terrestrial reference area in 2003 (Appendix A). The terrestrial reference area is located to the south of the DMTS, in the prevailing upwind location (AGRA 2001, Corps 2005) (Figure 3-2).

3.2.4 Streams

The following sections discuss the sediment and surface water data used to identify CoPCs in the stream environment, including site and reference stream data. 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 were available for a number of streams along the length of the DMTS road between the mine and the port (Figure 3-3). These included New Heart Creek, Aufeis Creek, Omikviorok River, and Anxiety Ridge Creek. Data were 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 were also available for downstream stations prior to confluence with other streams.

3.2.4.2 Reference Stream Sediment

Reference stream sediment samples were 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.

3.2.4.3 Site Stream Surface Water

Surface water data were available for a number of streams along the length of the DMTS road between the mine and the port (Figure 3-4). These included New Heart Creek, Aufeis Creek, Straight Creek, Omikviorok River, Mud Lake Creek, Tutak Creek, and Anxiety Ridge Creek. Data were 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 were also available for downstream stations prior to confluence with other streams.

3.2.4.4 Reference Stream Surface Water

Reference stream surface water samples were 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 the sediment and surface water data that were used to select CoPCs for tundra ponds, including site and reference data that were available at the time of the screening. 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 were available from four pond 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 were 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 were available from four pond 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 were 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 describe the sediment and surface water data, including site and reference data, that were used in the CoPC screening for the lagoon environment. 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 included 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 included samples at multiple stations in the Control Lagoon and the Reference Lagoon. The Control Lagoon and Reference Lagoon stations were 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 included samples at multiple stations in Ipiavik Lagoon, North Lagoon, Port Lagoon North, and Port Lagoon South.

3.2.6.4 Reference Lagoon Surface Water

Surface water data for the reference lagoons included samples at multiple stations in the Control Lagoon and the Reference Lagoon. The Control Lagoon and Reference Lagoon stations were 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 the sediment and surface water data used in the CoPC screening for the marine environment, including site and reference data. 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.

3.2.7.1 Site Marine Sediment

Marine sediment data for the site (Figure 3-3) included samples from a sampling grid in the nearshore area (located between 0 and 0.25 miles from shore, and up to 0.3 miles to the north and south of the port, centered on the shiploader area). Data were 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) were 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 included samples from stations in the nearshore area (located between 0 and 0.25 miles from shore, and up to 0.3 miles 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) were 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 0.10 level (alpha=0.10) to increase the likelihood of detecting differences (i.e., to increase the power of the test). The ANOVA method is more powerful than the non-parametric test, but underlying assumptions of equal variance and normality must be met. Method assumptions were evaluated using residual plots and normal probability plots. Even spread in the residual plots shows that the homogeneity of variance assumption was met and a straight line on a normal probability plot of the residuals indicates the normality assumption was 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. In cases where greater than or equal to 50 percent of site data values or 100 percent of the reference values were undetected, statistical analyses were not performed. Also, if the 90 percent confidence interval for the site mean concentration spanned zero because of small sample size and/or high variability, comparisons were not made. 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.

3.3 Human Health CoPC Screening

The human health CoPC screening was 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 were 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 are focused. In this investigation, chemicals present in ore concentrates were identified as site-related source materials and were 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 medium 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 affected 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 2003a) 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 medium, those chemicals that both exceeded their risk-based screening level *and* were significantly different from 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 (i.e., risk assessment) to determine if risks are elevated. In fact, because of the use of residential exposure assumptions, the use of maximum site concentrations, and the application of a 10-fold safety factor, the screening levels will tend to overestimate both exposure and toxicity and provide a very conservative approach to identifying CoPCs. Because of this, they are highly unlikely to mistakenly screen out chemicals that might be of concern and more likely to mistakenly identify CoPCs that are not present at concentrations of concern.

The remainder of this section describes the human health CoPC screening and the selection of CoPCs for each of the environments being evaluated: terrestrial, freshwater, coastal lagoon, and marine.

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 affected 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. The material site samples were composite samples collected from representative source material within each material site from beneath surface layers, where no prior excavation or exposure to dust deposition had occurred. Therefore, the chemical concentrations from these locations are considered representative of pre-mine or reference conditions for fill soils that were used to construct the road and facility areas. Thus, site (i.e., road and facility area) 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 (2003a) 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 noncarcinogens. A THQ of 0.1 results in a soil screening level for a chemical associated with a dose that is only one-tenth of the reference dose (RfD) or health-protective dose for that chemical. 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 (2003a) using the residential cleanup level formula and assumptions provided in DEC (2002), but with a target risk of 1×10^{-6} or a THQ of 0.1. For chemicals that are potentially carcinogenic, 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 risk 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

Most chemical-specific cancer slope factors (CSFs) and RfDs are provided in DEC (2002). For those chemicals not listed in DEC (2002), CSFs and/or RfDs were taken directly from U.S. EPA (2005). The cleanup level for lead listed in Table B1 of DEC (2003a) was not calculated using this methodology, but rather is the product of modeling using EPA's IEUBK child lead model. The IEUBK guidance (U.S. EPA 1994) 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, unlike for other chemicals, the screening level for lead is equivalent to the cleanup level.

DEC (2003a) guidance requires screening all sites using residential screening assumptions. Thus, site CoPCs were identified using residential exposure assumptions. However, there are no residences near the site and residential use is not expected in the future. In order to provide additional perspective on site concentrations based on the types 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 THQ of 0.1. These alternative non-residential screening levels are still a conservative means to evaluate the lower frequency exposures that might occur at the site 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), they incorporate a 10-fold safety factor, and they are still compared to maximum site concentrations. This non-residential 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:

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

where:

TR	=	target cancer risk 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

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:

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

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

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 residential risk-based screening levels: aluminum (2 of 51 samples exceeded), antimony (1 of 40), arsenic (54 of 75), barium (35 of 40), cadmium (236 of 478), iron (49 of 51), lead (279 of 479), manganese (37 of 40), thallium (1 of 12), and zinc (158 of 479). Only three chemicals (arsenic, cadmium, and lead) were present at concentrations exceeding non-residential risk-based screening levels. Arsenic exceeded the non-residential screening level in 1 of 75 samples, cadmium in 2 of 236 samples, and lead in 168 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 in road and facility areas 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 1 of 12 samples, and by less than 2-fold (maximum concentration of 1.32 mg/kg versus screening value of 0.9 mg/kg). In addition, the single thallium exceedance occurred in a road soil sample inside the ambient air boundary of the mine. Antimony exceeded the screening level in only 1 of 40 samples, and by less than 3-fold (maximum concentration of 14.8 mg/kg versus 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. Nevertheless, antimony and thallium were retained as human health CoPCs for the terrestrial environment in accordance with DEC (2002) CoPC screening procedures. Thus, antimony, barium, cadmium, lead, thallium, and zinc were retained as human health CoPCs for the terrestrial environment.

Sample screening for four of the 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). As described above, only one sample each for antimony and thallium exceeded residential screening criteria. Thus, figures illustrating spatial distribution for antimony and thallium were not prepared.

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 focuses on surface water nearest to the road, even though any potential fugitive dust-associated chemical concentrations in streams near the DMTS would be greatly diluted 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 (2003a) 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 noncarcinogens. 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 (2003a), using the residential cleanup level formula and assumptions provided in DEC (2002), but with a target risk of 1×10^{-6} or a THQ 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 risk 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:

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

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) to be 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 versus 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. Nevertheless, consistent with CoPC screening guidance (DEC 2002), thallium was retained as a human health CoPC for 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 were greater than the drinking water screening levels. Given that none of these four chemicals would be expected to bioaccumulate to a significant extent in edible fish tissues, screening criteria based on fish consumption would also be expected to be higher (i.e., less stringent) than drinking water screening levels if bioconcentration factors were available for the four chemicals to calculate them. Therefore, fluoride, molybdenum, and tin were not retained as CoPCs. However, lead was retained as a human health CoPC for the freshwater environment because of the uncertainty based on the lack of a fish consumption criterion and the fact that it is a primary CoPC for the site in the terrestrial environment. Thus, based on screening for both drinking water and fish consumption, lead and thallium were retained as human health CoPCs for 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 reference lagoon and marine water from areas not affected by the DMTS.

3.3.3.1 Comparison of Site Lagoon and Marine Data with Reference Data

Site lagoon and marine water and sediment chemical concentrations were compared to reference data to determine which constituents have elevated concentrations at the site.

3.3.3.1.1 Lagoon Environment

As described in Section 3.2.6, reference 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; Corps 2005) of the port site facilities. The results of the lagoon water reference comparison, summarized in Table 3-11, indicate that antimony, fluoride, lead, and molybdenum are statistically elevated compared to reference conditions. A statistical comparison to reference could not be made for mercury, selenium, tin, or vanadium, because there were too few detections in site and reference data. Mercury was not detected in any site or reference sample (Table 3-17). A statistical comparison to reference samples could not be made for tin because it was detected in only one of eight site samples and was not detected in any reference sample. Selenium and vanadium were both detected in five of eight site samples, and were undetected in reference samples. For all other CoPCs, site concentrations were not statistically elevated compared to reference conditions (Table 3-1).

3.3.3.1.2 Marine Environment

As described in Section 3.2.7, marine water and sediment 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). The results of the marine water reference comparison, as summarized in Table 3-13, indicate that only selenium, silver, and strontium are statistically elevated compared to reference concentrations. Chromium, mercury, nickel, and zinc were not detected in any site sample (Table 3-18). Statistical comparisons to reference samples could not be made for copper, thallium, tin, and vanadium because sample results in site and reference areas were mostly undetected. 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 (Table 3-18). Thus, selenium, silver, strontium, copper, and thallium cannot be screened out by comparison with reference conditions.

The results of the marine sediment comparison, summarized in Table 3-12, indicate that barium, cadmium, chromium, copper, silver, strontium, and zinc are statistically elevated over reference conditions. Statistical comparisons to reference samples could not be made for antimony, mercury, selenium, and tin because sample results in site and reference areas were mostly undetected. For lead, there was insufficient statistical power to distinguish the mean site concentration from zero (and therefore insufficient power to distinguish it from the reference mean), because of the high variability in lead concentrations. Therefore, a statistical comparison with reference was not made for lead.

Note that the comparison of site and reference marine sediments described in the preceding paragraph was done with data collected prior to 2004 (data used for screening were described in Section 3.2). However, as agreed upon with DEC, supplemental sediment samples were collected in 2004 from the shiploader area and analyzed for CoPCs as part of the Phase II field sampling and analysis program for the DMTS risk assessment (see Section 4). These data were used to assess current conditions a year after completion of additional shiploader and barge dust controls. The first of two sampling events was conducted in early June 2004, prior to the start of shipping activities at the port site, and the second was conducted during the shipping season (September 2004). All concentrations were below screening criteria for all samples from both sampling events (pre-shipping and during-shipping) in 2004, and thus a site/reference comparison was not relied upon for CoPC screening. Section 4 describes the sampling and provides the 2004 sample results in comparison to screening criteria.

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.

As described in the CSM, people would come into little or no direct contact with lagoon or 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. There are no complete exposure pathways in the lagoon environment because there are no fish or shellfish collected for human consumption. The primary potential exposure pathway in the 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 sediment quality standards (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).

3.3.3.2.1 Lagoon Environment

In lagoon water, all arsenic samples from both the site and reference lagoons exceeded the AWQC and WDOE surface water criterion (Table 3-17). No other CoPC exceeded its AWQC or WDOE surface water criterion (Table 3-17).

3.3.3.2.2 Marine Environment

In marine water, arsenic exceeded its AWQC or WDOE surface water criterion in seven of nine site samples and five of seven reference samples (Table 3-18). No other CoPC exceeded its AWQC or WDOE surface water criterion (Table 3-18).

For marine sediment, Section 3.5.5.1 presents a comparison of sediment data with ecologically based screening levels. When marine sediment chemical concentrations were compared to SQS, all chemicals in the marine environment were screened out except 1 of 136 cadmium samples and 3 of 136 zinc samples. However, he maximum zinc concentration in marine sediments (2,550 mg/kg) was still lower than the residential soil screening criteria for zinc of 4,100 mg/kg (described in Section 3.3.1.2). Thus, even with the higher direct contact assumed in the soil screening criteria, human exposure to the zinc concentrations in marine sediments would not pose a risk to human health. The single cadmium sample exceeding the SQS had a concentration of 52.9 mg/kg.

Note that the screening of marine sediments described in the preceding paragraph was done with data collected prior to 2004 (data used for screening were described in Section 3.2). However, as agreed upon with DEC, supplemental sediment samples were collected in 2004 from the shiploader area and analyzed for CoPCs as part of the Phase II field sampling and analysis program for the DMTS risk assessment (see Section 4). These data were used to assess current conditions a year after completion of additional shiploader and barge dust controls. The first of two sampling events was conducted in early June 2004, prior to the start of shipping activities at the port site, and the second was conducted during the shipping season (September 2004). All concentrations were below screening criteria for all samples from both sampling events (pre-shipping and during-shipping) in 2004, and thus a site/reference comparison was not relied upon for CoPC screening. Section 4 describes the sampling and provides the 2004 sample results in comparison to screening criteria.

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

3.3.3.3.1 Lagoon Environment

There were no CoPCs in lagoon water with a maximum site concentration that exceeded both the reference concentrations and risk-based screening levels. Thus, even if there were complete exposure pathways for lagoons, there are no lagoon water CoPCs. As discussed in Section 2.3.3.3, the lagoon environment near the DMTS is not evaluated further in the HHRA because 1) it is not used for subsistence fish or shellfish collection, and 2) people do not have an appreciable amount of direct contact with site lagoon water or sediments.

3.3.3.3.2 Marine Environment

There were no chemicals in marine water with a maximum site concentration that exceeded both the reference concentrations and their risk-based screening levels (see Section 3.3.3.2.2). Thus, there are no marine water CoPCs. In the supplemental 2004 marine sediment sampling program (described in Section 4), all CoPC concentrations were below screening criteria for all sediment samples from both sampling events (pre-shipping and during-shipping) in 2004. Thus, there are no CoPCs for the marine environment and thus it will not be evaluated further in the risk assessment.

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: antimony, barium, cadmium, lead, thallium, zinc
- Freshwater environment: lead, thallium
- Lagoon 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 presented in this section 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 in the screening assessment, then further assessment is conducted in the baseline ERA (Section 6) 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, and identify the CoPCs and receptors to be assessed quantitatively in the ERA (which is presented in Section 6 of this document).

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). There are very few screening benchmarks available for nonvascular plants and therefore they were not used in the ecological screening assessment. 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. For all chemicals that have ORNL phytotoxicity benchmarks, maximum measured concentrations for these chemicals 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 exceeded their TECs only 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 criteria 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 the range of freshwater criteria concentrations that were calculated using the minimum and maximum hardness values for stream surface water. EPA also provides a default water quality criterion based on a hardness of 100 mg/L $CaCO_3$. This value is presented in parentheses in Table 3-21 as well. Table 3-21 presents freshwater AWQC 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 to 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. (These criteria are described above in Section 3.5.4.1.) 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).

The elevated cadmium concentrations depicted in Figure 3-9 were all collected during one sampling event in August 2000 (Corps 2001). Although no error is apparent in the quality assurance documentation for this event, these results were inconsistently high compared with results from multiple sampling events before and after this event. Additional sediment samples were collected as part of the Phase II field sampling analysis program to characterize current conditions. The Phase II sediment sampling results are summarized in Section 4.0.

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) was 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, were used 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 was 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 assumed 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 were used in the exposure estimates (data tables were included in Appendix C). These conservative assumptions represented a worst-case exposure scenario;

thus, using these values resulted in protective exposure estimates that were 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 would be 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 were compared to no-observed-adverse-effect level (NOAEL) TRVs to calculate hazard quotients. For screening purposes, if the ratio of exposure to the TRV was less than or equal to 1.0, then the chemical was not considered likely to cause adverse effects to upper-trophic-level receptors, and was not retained as a CoPC. Chemicals that had hazard quotients greater than 1.0 in these conservative food-web models were 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 had been analyzed for CoPCs at the time of the CoPC screening were several plant species (moss, lichen, willow, berries); therefore, data were only available to directly evaluate exposure to herbivorous receptors. However, because of the elevated chemical concentrations in plants, particularly moss, exposure of herbivorous wildlife likely represents one of the most important exposure pathways. The tundra vole was 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 had been analyzed for the broadest range of chemicals, and for those chemicals that had been measured in more than one plant type (i.e., lead, zinc, cadmium), the maximum concentrations in moss were 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 were not necessarily collocated 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 had 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 could not 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 because 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 was substantially lower than the maximum site hazard quotient, indicating that there are potentially incremental risks to receptors resulting from exposure to these four chemicals at the site. However, because the hazard quotient for manganese was 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 were 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 were limited to four chemicals: cadmium, lead, selenium, and zinc, so only these four chemicals could be analyzed in the screening models (Morris and Ott 2001; Appendix C).

Fish data were used to model exposure for two piscivorous receptors: red-throated loon and river otter. Exposure parameters for these two receptors that were 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 red-throated loon are shown in Tables 3-31 and 3-32, respectively. For river otter, all hazard quotients were less than or equal to 1.0, while for loons, hazard quotients for lead, zinc, and cadmium were less than 1.0, but the selenium hazard quotient was 1.2 based on fish data from Aufeis Creek. Although the selenium hazard quotient equaled 1.0 for river otter and slightly exceeded 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 indicated a low likelihood of unacceptable risk to piscivorous wildlife from exposure to cadmium, lead, selenium, and zinc, and further evaluation of risk to piscivorous wildlife foraging in freshwater streams and creeks is not required.

3.5.6.3 Invertivorous Wildlife

Effects to benthic invertivores that may forage in freshwater or coastal marine habitats could not be assessed directly, as no data had been collected on chemical concentrations in benthic invertebrates at the time of the CoPC screening. However, chemical data were 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, the 90th percentile biota-sediment accumulation factors (BSAFs) from Bechtel Jacobs (1998) were used in exposure models for arsenic, cadmium, chromium, copper, mercury, nickel, lead, and zinc to estimate metals concentrations in invertebrate prey based on available sediment data. These BSAF values were 0.69, 7.99, 0.468, 5.25, 2.868, 2.32, 0.607 and 7.527, respectively. For other metals, a BSAF of 1.0 was used in the exposure models. A similar approach was used for estimating background risk based on maximum chemical concentrations measured in reference creeks, ponds, and lagoons. Estimated benthic invertebrate concentrations were used to model exposure to the common snipe, which was selected as the representative freshwater invertivorous species (creeks and tundra ponds) and the black-bellied plover, which was 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 could not be calculated for these chemicals. In all creeks and streams traversed by the haul road, chemicals with hazard quotients exceeding 1.0 using the conservative estimate of benthic invertebrate tissue concentrations included aluminum, barium, chromium, and zinc. Cadmium, lead, and mercury hazard quotients in Anxiety Ridge Creek also exceeded 1.0 (Table 3-33). However, hazard quotients for aluminum, barium, and chromium also exceeded 1.0 in the reference creek, and the site and reference hazard quotients differed by less than 2-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 were less than those calculated at the reference lagoons, while the hazard quotient for selenium was less than 2-fold greater than the corresponding reference area hazard quotient. Seven chemicals had hazard quotients exceeding 1.0 in coastal lagoons: aluminum, barium, cadmium, chromium, lead, mercury, and zinc (Table 3-35). However, all of these chemicals except cadmium, lead, and mercury also had hazard quotients equal to or exceeding 1.0 in the reference lagoons. Hazard quotients for aluminum, barium, and chromium in site and reference lagoons were almost equal. The hazard quotients in site lagoons for cadmium, lead, and zinc were more than 2-fold greater than the corresponding reference area hazard quotients.

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 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 the 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 used a two-tiered approach to identify which chemicals should be retained as CoPCs and which ones could be eliminated from further consideration. In the first tier, maximum chemical concentrations in each environmental medium were compared with the lowest available screening benchmarks. In the second tier, the statistical comparisons with reference area concentrations were performed. At the first tier, chemicals that passed (i.e., maximum concentrations were lower than a screening value) were dropped from the evaluation, while chemicals that failed the comparison were carried forward to the next tier. Chemicals that were undetected in all samples were eliminated if concentrations were less than screening values when their value was expressed as one-half of the detection limit, but were retained otherwise. Chemicals with no appropriate screening benchmarks in a specific medium were carried forward to the next tier.

3.6.1.1 First Tier Media Screening

In the first tier, media concentrations were 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 summarized in Table 3-36 (benchmark comparisons are presented by medium and environment in Tables 3-19 to 3-27). 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

In the second comparison, all chemicals remaining after the first 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 summarized in Table 3-37 (statistical comparisons of site and reference data are presented by medium and environment in Tables 3-5 to 3-13), and discussion of the reference area selection is provided in Section 6.6. The following chemicals were screened out at this tier:

- Aluminum, chromium, iron, and nickel in tundra soil
- Barium and vanadium in stream sediment
- Aluminum, cobalt, iron, and strontium in stream water
- Barium, copper, nickel, selenium, thallium, and vanadium in tundra pond sediment
- Aluminum, barium, cobalt, copper, fluoride, iron, lead, molybdenum, strontium, and vanadium in tundra pond water
- Aluminum, barium, cobalt, iron, molybdenum, nickel, selenium, strontium, thallium, and vanadium in lagoon sediment
- Aluminum, arsenic, barium, cobalt, iron, nickel, strontium, and zinc in lagoon water
- Aluminum, arsenic, cobalt, fluoride, iron, manganese, molybdenum, nickel, thallium, and vanadium in marine sediment
- Aluminum, antimony, barium, cobalt, fluoride, iron, manganese, and molybdenum 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 3-38. 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 resulting from 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 were retained as CoPCs for the baseline assessment, risk characterization is 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

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

3.6.2.2 CoPCs in Stream Habitats

Five chemicals (arsenic, cadmium, lead, nickel, and zinc) in stream sediment failed the screening based on comparisons with benchmarks and reference area concentrations and were retained as CoPCs for the baseline ERA. Nine other chemicals (antimony, cobalt, fluoride, molybdenum, selenium, silver, strontium, thallium, and tin) lack appropriate sediment screening benchmarks and were not screened out based on comparisons with reference area concentrations, and were retained on this basis. Three chemicals (cadmium, lead, and zinc) in stream water failed the screening based on comparisons with benchmarks and reference concentrations. However, eight chemicals that lack benchmarks (antimony, barium, fluoride, manganese, molybdenum, thallium, tin, and vanadium) were not screened out based on comparisons with reference stream data and were 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 were retained as CoPCs for the baseline ERA, as were seven other chemicals (antimony, cobalt, fluoride, molybdenum, silver, strontium, and tin) that lack relevant sediment screening benchmarks and were not screened out based on statistical comparisons with reference data.

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 were retained as CoPCs for the baseline ERA. Four additional chemicals (antimony, manganese, thallium, and tin) that lack AWQC and were not screened out based on comparisons with reference concentrations were retained on this basis.

3.6.2.4 CoPCs in Coastal Lagoon Habitats

Four chemicals (arsenic, cadmium, lead, and zinc) in lagoon sediment failed the screening based on comparisons with benchmarks and reference area concentrations, and were retained as

CoPCs for the baseline ERA. Four chemicals in lagoon sediments (antimony, fluoride, manganese, and tin) and seven chemicals in lagoon water (antimony, fluoride, manganese, molybdenum, thallium, tin, and vanadium) lack appropriate screening benchmarks and were not screened out based on comparisons with reference area concentrations.

3.6.2.5 CoPCs in Marine Habitats

Six chemicals (cadmium, copper, lead, mercury, silver, and zinc) in marine sediment failed the screening based on comparisons with benchmarks and reference area concentrations. 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.

Copper is the only chemical in marine water that failed the screening based on comparisons with benchmarks and reference area concentrations. Four chemicals lack appropriate benchmarks (strontium, 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.

The initial screening of marine sediments and surface water was performed using data collected prior to 2004 (data used for screening were described in Section 3.2). However, as agreed upon with DEC, supplemental sediment samples were collected in 2004 from the shiploader area and analyzed for CoPCs as part of the Phase II field sampling and analysis program for the DMTS risk assessment (see Section 4). These data were used to assess current conditions a year after completion of additional shiploader and barge dust controls. The first of two sampling events was conducted in early June 2004, prior to the start of shipping activities at the port site, and the second was conducted during the shipping season (September 2004). All sediment concentrations were below ecological screening benchmarks for all samples from both sampling events (pre-shipping and during-shipping) in 2004, and thus a site/reference comparison was not relied upon for CoPC screening. Section 4 describes the sampling and provides the 2004 sample results in comparison to screening criteria.

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 (described in Section 3.5.6). 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 red-throated 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 could not 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 was 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 was not retained as a CoPC for terrestrial herbivores. The 14 chemicals that were retained by the screening exercise are evaluated in the baseline ERA (Section 6) 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 could not be screened out, but they also cannot be evaluated quantitatively in exposure models. These two chemicals are therefore evaluated qualitatively in the baseline ERA, where the likelihood of risk from these chemicals is discussed relative to risk from chemicals for which derivation of numeric hazard quotients is possible. Screening was not performed for terrestrial carnivores or terrestrial insectivores, because of data gaps for chemical concentrations in prey of these receptors. Therefore, the same suite of chemicals identified as CoPCs for terrestrial herbivores were also evaluated for risk to carnivores and insectivores, by collection of appropriate prey species and analysis for chemicals concentrations during the supplemental field sampling program (described in Section 4).

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 as 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. Because CoPCs have not been measured in fish from coastal lagoons, the supplemental sampling program (Section 4) included attempted fish collection and analysis to assess risk to piscivorous wildlife potentially using those lagoons.

Exposure models for benthivorous birds indicate that cadmium, lead, mercury, and zinc require further evaluation in freshwater creeks and streams, as the hazard quotients for these chemicals were greater than 1.0 and were at least 2-fold higher than reference hazard quotients in one or more site streams. The hazard quotient for selenium equaled 1.0 in Aufeis Creek, 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 exceeded 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 2-fold, indicating that the incremental risk as a result of exposure to these chemicals at the site is minimal. Therefore, these three chemicals were not retained as CoPCs for benthic invertivores in freshwater creeks and streams.

In tundra ponds, nine chemicals had hazard quotients exceeding 1.0, but only five chemicals had hazard quotients that were also more than 2-fold higher than the corresponding reference pond hazard quotients. These five chemicals, cadmium, lead, mercury, thallium, and zinc, were retained as CoPCs for benthic invertivores in tundra ponds.

In coastal lagoons, seven chemicals had hazard quotients greater than 1.0. However, only hazard quotients for cadmium, lead, and zinc were more than 2-fold higher than the corresponding reference hazard quotient. Therefore, these chemicals were 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 could not 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 are therefore evaluated qualitatively in the baseline ERA (Section 6), where the likelihood of risk from these chemicals is discussed relative to risk from chemicals for which derivation of numeric hazard quotients is possible.

No data were available to evaluate potential effects on herbivorous wildlife that may feed on aquatic plants in freshwater or coastal lagoon habitats. To address this data gap, plants were 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 (see supplemental data collection described in Section 4).

In summary, CoPCs that are retained for evaluation in quantitative food-web models for higher trophic-level wildlife in the baseline ERA 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 piscivorous wildlife foraging in freshwater streams and creeks: no CoPCs identified
- For avian invertivores foraging in freshwater streams and creeks: cadmium, lead, mercury, and zinc
- For avian invertivores foraging in tundra ponds: cadmium, lead, mercury, thallium, and zinc
- For avian invertivores foraging in coastal lagoons: cadmium, lead, and zinc.

3.7 Data Gaps

There were sufficient data for completion of the CoPC screening in primary media. As shown in Table 3-3, there were at least 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. These data were also used in the risk assessment, as described in Sections 5 and 6. The results of the

CoPC screening analyses (described in Sections 3.3 through 3.6) helped to identify additional data needs. The most significant data gaps were for biological media, both for the human health and the ERAs.

The ERA required biota sample collection to obtain data for food or prey items associated with the receptors. Table 3-39 summarizes the ERA data needs in relation to each environment, assessment endpoint, receptor, and associated food item. This table was Table 1 from the field sampling and analysis plan (Exponent 2004a). This sampling program is discussed in the following section.

Additional data needs for the HHRA included the collection of ptarmigan (a subsistence food item) to be analyzed for antimony, barium, cadmium, lead, thallium, and zinc. Additional berry and sourdock data were also needed because earlier sampling programs did not include all of these CoPCs as analytes.

Further details on the collection and analysis of samples to address the HHRA and ERA data needs are provided in Section 4, which describes the Phase II data collection conducted in summer 2004.

4 Supplemental Data Collection for Risk Assessment

The Phase II field study for the DMTS fugitive dust risk assessment was conducted in summer 2004 to provide additional biota data needed to assess possible risk to human health and the environment from fugitive dust deposition. Specific data needs for the ERA and HHRA were discussed in the prior section (Section 3.7) and ERA data needs were also outlined in Table 3-39.

The field program included assessments in terrestrial, freshwater aquatic (stream and tundra pond), coastal lagoon, and marine environments. Biota samples included small mammals, ptarmigan, terrestrial and aquatic invertebrates, berries, and other vegetation, in which CoPC concentrations were analyzed. Media associated with these biota, including tundra soil and sediment, were also sampled. The health of aquatic invertebrate communities and plant communities was also assessed. Water quality measurements (pH, dissolved oxygen, temperature, conductivity, and salinity) were collected in streams, ponds, and lagoons, and pH was also measured in tundra soils.

The locations of stations sampled during the Phase II sampling event are illustrated in Figures 4-1 through 4-5 and the schematic layouts of typical terrestrial transect stations are provided in Figures 4-6 and 4-7. A typical stream station is illustrated in Figure 4-8. Table 4-1 provides an overview of the number of stations and media sampled is provided in Table 4-1, and Table 4-2 provides additional detail about the samples and the analyses conducted.

The following subsections provide a brief overview of the Phase II sampling program. The terrestrial, freshwater aquatic, coastal lagoon, and marine assessments are discussed in the following three subsections: Section 4.1, *Human Health–Subsistence Foods Data*; Section 4.2, *Ecological Data*; and Section 4.3, *Marine Assessment and CoPC Screening* (which is relevant to both the HHRA and ERA). A brief description of the nature of the data collection in each environment is provided. For further detail, Appendix E provides a discussion of the sampling activities conducted during the Phase II program, including field modifications relative to the sampling and analysis plan (Exponent 2004a). Data tables for the Phase II program are included in Appendix G. Photographs of typical sample media are provided in Appendix J.

4.1 Human Health—Subsistence Foods Data

Supplemental subsistence foods data collected in 2004 for use in the HHRA include salmonberries, sourdock, and ptarmigan. Detailed discussion of the sample collection and any field modifications is included in Appendix E, and data tables are included in Appendix G. Appendix H provides detailed discussion in several technical memoranda reviewing subsistence foods data, including one technical memorandum reviewing berry and sourdock data, another reviewing ptarmigan data, and a third reviewing available caribou data. These data are discussed further and summarized for use in the HHRA in Section 5.

4.2 Ecological Data

Supplemental data collected in 2004 for use in the ERA included data from the terrestrial, freshwater aquatic, and coastal lagoon assessments. A brief description of the nature of the data collection in each environment is provided. Detailed analysis of the ecological data is provided in Section 6.

4.2.1 Terrestrial Assessment

The terrestrial assessment included collection of CoPC concentration data for small mammals, soil invertebrates, vegetation tissue, and tundra soil in areas surrounding the DMTS and mine. In addition, vegetation community analyses were performed. The sampling locations are illustrated in Figure 4-1, and a summary of the sample types collected at each station is provided in Tables 4-1 and 4-2. Typical station layout is illustrated in Figures 4-6 and 4-7. Detailed discussion of the sample collection and any field modifications is included in Appendix E, and data tables are included in Appendix G. Vegetation community data and survey narratives are included in Appendix I. Detailed analysis and discussion of the terrestrial assessment data is provided in Section 6.

An overview of the tundra soil and biota tissue data is provided in Figures 4-9 through 4-12, which illustrate barium, cadmium, lead, and zinc concentrations, respectively. Tissue concentrations are plotted on a dry weight basis, which is how they were reported by the analytical laboratory. Each of the small plots within a figure shows concentration versus distance away from the DMTS road or facilities on a given terrestrial transect. Each column of plots within the figure reflects the tundra soil and biota tissue concentration data available for a given terrestrial transect, beginning at the port on the left, traversing the road across the middle, and ending at the mine on the second column from the right. The right-most column of plots is the terrestrial reference area, where the concentrations are plotted by station number rather than distance.

Cadmium, lead, and zinc (Figures 4-10 through 4-12) were plotted because they have been a common focus of the characterization studies, as important constituents of ore concentrates and fugitive dust, and as potential risk drivers. Concentrations in all of the media sampled typically decrease with distance away from the DMTS road, port facilities, and the mine ambient air boundary. Along the length of the DMTS transportation corridor, higher concentrations of these three metals generally occur at each end of the road and lower concentrations generally occur in the central portion of the road. This is the result of the tracking of concentrates that has occurred from the mine and port concentrate loading and unloading facilities over time. These patterns have been observed with characterization data from previous field programs that were mapped in earlier documents, such as the 2001 fugitive dust data report (Exponent 2002a), the fugitive dust background document (DEC et al. 2002), and the port site characterization report (Exponent 2003c).

Barium (Figure 4-9) was plotted to illustrate the different pattern that occurs in comparison with cadmium, lead, and zinc. Ore and waste rock are rich in barium, but the ore concentrate is not. As a result, barium concentrations are higher on the two transects near the mine than they are on the other transects. However, as with other CoPCs, barium concentrations decrease with distance away from the DMTS road, port facilities, and the mine ambient air boundary.

Hydrogen potential (pH) measurements were also made on tundra soil samples at each station (tabulated in Appendix G). A trend of decreasing pH versus distance from the DMTS road and port facilities was apparent. At the 1,000-m stations, the pH was similar to reference pH values. Noting that the pH scale is logarithmic, there is a difference of approximately three orders of magnitude in hydrogen ion concentrations ($[H^+]=1/10^{pH}$) over the length of the 1,000-m transect, as compared with a two order of magnitude difference in metals concentrations. Figure 4-13(a) illustrates the pH and lead trends in tundra soil samples along terrestrial transect TT8, located in the middle portion of the DMTS road. Between the road and the 400-m station, pH varied within the range of 6.9 to 7.7. Beyond the 400-m station, pH first declined below 6.0 at the 600-m station, declined below 5.0 at the 750-m station, and reached the upper end of the reference range (3.9–4.5) at the 1,000-m station. Figure 4-13(b) illustrates pH along with several additional metals on a normalized scale, indicating similar trends among the metals. Figure 4-13 also shows that metals concentrations decrease more rapidly than pH with distance from the DMTS road. This phenomenon is likely a result of the alkaline nature of dust emanating from the road, which includes dust from calcareous rock (used to construct or maintain some portions of the road), as well as calcium chloride, which is applied as a hygroscopic dust control agent. There may also be a secondary effect resulting from a decline in sphagnum mosses, which tend to acidify their environment. Further discussion of these trends and factors is included in the terrestrial plant community analysis in Section 6.2, particularly Section 6.2.3.1.

4.2.2 Freshwater Aquatic Assessment

The freshwater aquatic assessment included collection of CoPC concentration data in biota (aquatic invertebrate whole body tissue, and vegetation tissue) at streams and tundra ponds near the DMTS and streams and tundra ponds in the reference area. Aquatic invertebrate community samples were collected at these stations, along with water quality measurements. Sediment samples were also collected from streams, and tundra soil was collected adjacent to streams and ponds at these stations. The sampling locations are illustrated in Figure 4-3, a typical stream station is illustrated in Figure 4-8, and a summary of the sample types collected at each station is provided in Tables 4-1 and 4-2. Detailed discussion of the sample collection and any field modifications is included in Appendix E, and data tables are included in Appendix G. Detailed analysis and discussion of the freshwater aquatic assessment data is provided in Section 6.

4.2.3 Coastal Lagoon Assessment

The coastal lagoon assessment included collection of CoPC concentration data in sediment and biota (aquatic invertebrate tissue, and vegetation tissue) in site and reference lagoons. Aquatic invertebrate community samples were collected at the lagoon stations, and extra sediment volume was collected for sediment toxicity testing for invertebrates. The invertebrate community samples were archived pending the results of the sediment toxicity testing. Taxonomic analysis of the invertebrate community samples was ultimately not conducted, as the toxicity testing results indicated that effects to the invertebrate community are unlikely (see discussion in Section 6.4.1). Fish sampling was also attempted in the coastal lagoons. Water quality measurements were made at the lagoon stations, and tundra soil samples were collected at these stations. As in the terrestrial environment, vegetation community analyses were also performed at site and reference lagoon stations. The sampling locations are illustrated in Figure 4-4, and a summary of the sample types collected at each station is provided in Tables 4-1 and 4-2. Detailed discussion of the sample collection and any field modifications is included in Appendix E, and data tables are included in Appendix G. Vegetation community data and survey narratives are included in Appendix I. Detailed analysis and discussion of the lagoon assessment data is provided in Section 6.

4.3 Marine Assessment and CoPC Screening

The purpose of the marine assessment was to evaluate current CoPC concentrations in surface sediments at stations in the Chukchi Sea in the vicinity of the shiploader, one year after major shiploader and lightering barge improvements were made to further control fugitive concentrate dust. (The shiploader and barge improvements were completed in June 2003). The station locations (Figure 4-5) were selected primarily on the basis of historical evaluations (RWJ 1997; Exponent 2003d) and offshore current patterns (prevailing current is northward), and were designed to allow evaluation of gradients of CoPC concentrations in relation to sources, as well as temporal changes in CoPC concentrations (i.e., by resampling stations from previous studies). A summary of the sample types collected at each station is provided in Tables 4-1 and 4-2. The following modifications were made to the Phase II sampling strategy for the June 2004 marine assessment outlined in Exponent (2004a):

- A modified Ponar grab sampler was used to collect sediment samples rather than the stainless-steel Ekman grab sampler, modified petite-Ponar grab sampler, or a DRCV corer suggested in Exponent (2004a). The modified Ponar grab sampler provides the same quality of sediment sample, but the grab sampler is slightly larger than the petite version and therefore provides more sediment per grab.

- The location of Station NM-REF-1 was adjusted slightly to match the station coordinate sampled during the 2003 and June 2004 sampling events. Station NM-REF-1 was placed as close as possible to the beach and the previously sampled station coordinate.

The quality and usability of the data generated from this field event were not affected by these modifications. Detailed discussion of the sample collection is included in Appendix E, and data tables are included in Appendix G.

Sediment samples were collected in two sampling events to evaluate possible seasonal variability in exposures in the marine environment. The first event was conducted in early June 2004, prior to the start of shipping activities at the port site, and the second was conducted during the shipping season (September 2004).

The stations that were sampled (see Figure 4-5) are located on a grid that has been sampled historically in the vicinity of the port site (RWJ 1997; Exponent 2003c). Chemicals that had concentrations in exceedance of the marine screening benchmarks and that were higher than reference concentrations in 2003 (cadmium, copper, lead, mercury, silver, and zinc) were analyzed again in 2004 at a subset of 7 of the 26 grid stations (see Figure 4-5). Lead, zinc, and cadmium analyses were conducted at all of the remaining grid stations. The subset of seven locations (NMD, NMGZ, NML, NMM, NMN, NMO, and NMAA) included the 4 stations where these chemicals exceeded benchmarks in 2003 (i.e., NMD, NMGZ, NML, and NMM), and also represented a range of concentrations observed historically, based on data collected previously (RWJ 1997; Exponent 2003e).

Reference site samples were collected from three stations at an area approximately 4 km south (upcurrent) of the port site facilities (see Figure 4-5). The three reference locations selected (NM-REF-1, NM-REF-2, and NM-REF-3) have grain size composition similar to the onsite stations.

The analytical results were compared with ERL and ERM guideline values developed by Long et al. (1995) for marine sediment and with the Washington State SQS (WAC 173–204).[10] The results are shown in comparison with the screening criteria in Figures 4-14 through 4-19 for the 2004 pre-shipping event, and in Figures 4-20 through 4-25 for the 2004 during-shipping event. The concentrations were below all of the screening criteria for all samples from both sampling events (pre-shipping and during-shipping) in 2004. As a result, no CoPCs were identified for the marine environment.

[10] These are ecologically-based screening criteria that are described in further detail in Section 3.5.4.1. There are, unfortunately, no sediment screening criteria available that are specifically derived to be protective of human health. However, criteria that are conservatively protective of aquatic life are likely to also be protective of human health.

5 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 and 3, respectively. The following sections describe the methodology and the results of the DMTS HHRA. Section 5.1 describes refinements to the preliminary CSM based on the results of the CoPC screening. Section 5.2 presents the methodology used in the exposure assessment, which quantifies the amount of exposure to site CoPCs that could potentially occur. Section 5.3, the toxicity assessment, summarizes current scientific knowledge regarding the toxicity of site CoPCs. Section 5.4, the risk characterization, combines the results of the exposure and toxicity assessments to derive risk estimates for the site, interprets the risk estimates, and discusses uncertainties in the risk assessment.

5.1 Refined Conceptual Site Model

Based on the results of the human health CoPC screening in Section 3 and the screening of supplemental marine sediment data in Section 4, there are six CoPCs in the terrestrial environment (antimony, barium, cadmium, lead, thallium, and zinc, two CoPCs in the freshwater environment (lead and thallium), and no CoPCs in the marine environment. Potential exposures related to the marine environment were not evaluated in the risk assessment because the conservative screening process indicated that there is little or no risk related to site activities in the marine environment. The refined CSM (Figure 5-1) reflects the results of the screening process. The exposure pathways in the terrestrial and freshwater environments remain unchanged from the preliminary CSM. Thus, risks were quantitatively evaluated for soil and dust ingestion, water ingestion, and subsistence food consumption (Figure 5-2) in the terrestrial environment. In the freshwater environment, risks were quantitatively evaluated for water ingestion and subsistence fish consumption.

5.2 Exposure Assessment

In a 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.

5.2.1 Exposure Concentrations

EPA guidance (U.S. EPA 1989, 1992b, 2002b) indicates that exposure point concentrations (EPCs) used in risk assessment calculations should be either the 95 percent upper confidence limit (95%UCL) on the mean concentration or the maximum site concentration, whichever is lower. EPA recommends the 95%UCL as an estimate of mean exposure concentration because of the uncertainty associated with estimating the true average exposure concentration at a site.

Each site data set was tested for whether it fit a normal, gamma, or lognormal distribution. If these distributions fit, then the appropriate 95%UCL for the fitted distribution was used. If none of these distributions fit, then a non-parametric UCL was used. All UCL calculations were conducted using EPA's ProUCL 3.0 Software, in accordance with EPA exposure point guidance (U.S. EPA 2002b,d). If the 95%UCL on the mean was greater than the maximum value, the maximum concentration was used instead.

EPCs for lead were calculated using arithmetic means. As described below and in model guidance, the IEUBK and ALMs are designed to be applied using average values as input. A geometric standard deviation (GSD) for blood lead values in the general population is then applied to account for variability.

5.2.1.1 Exposure Point Concentrations for Environmental Media

EPCs for water and soil are presented in Table 5-1. DMTS soil data were available for the road, road shoulders, and the port area. Because concentrations differ significantly between the port area and the remainder of the DMTS, while exposure is assumed to occur randomly throughout the entire area, area-weighted-exposure concentrations were calculated for use in the risk assessment. To do this, it was necessary to make assumptions about the extent of area that could be represented by soil concentration data from samples taken from the road and road shoulder, and samples taken from the port facilities area. Although it has been demonstrated that concentrations decrease significantly within 1 km from the DMTS, for the purpose of the HHRA it was conservatively assumed that site soil concentrations are representative of conditions as far as 5 km downwind and 2 km upwind of the DMTS road and ambient air boundaries.

Figure 5-3 shows the geographic area identified as the subsistence use area for Kivalina residents (as reported in Dames & Moore 1983a). Within the Kivalina subsistence use area, the assumed site area represented by port soil concentrations (on the port side of the ambient air boundary) measures 3,759 hectares.[11] The assumed site area represented by road and road shoulder soil concentrations (on the mine side of the port ambient air boundary) measures 44,858 hectares. Thus, the total assumed site area represented by soil concentrations is 48,617 hectares, of which 8 percent is port area (i.e., 3,759 hectares/48,617 hectares = 0.08) and 92 percent is road area (i.e., 44,858 hectares/48,617 hectares = 0.92). The DMTS area-weighted soil EPCs were, thus, calculated using the following formula:

$$\text{DMTS Area-weighted Soil EPC} = (\text{Port Soil EPC} \times 0.08) + (\text{Road Soil EPC} \times 0.92)$$

Figure 5-4 shows the geographic area identified as the subsistence use area for Noatak residents (as reported in Dames & Moore 1983a). Within the Noatak subsistence use area, the assumed site area represented by port soil concentrations (on the port side of the ambient air boundary) also measures 3,759 hectares. However, for Noatak the entire DMTS and the mine reside within the subsistence use area. Thus, the port area, where concentrations are higher, comprises a smaller portion of the total DMTS area (3,759 out of 69,725 hectares, or 5 percent). This would result in lower area-weighted concentrations. Thus, the more conservative value produced by using the Kivalina subsistence use area was used to derive area-weighted soil EPCs (Table 5-1).

At the request of DEC, soil EPCs were also derived using an area-averaging approach, calculated as the arithmetic average of the Port Soil EPC and Road Soil EPC.

5.2.1.2 Exposure Point Concentrations for Subsistence Foods

EPCs for fish, caribou, ptarmigan, salmonberries, and sourdock are presented in Table 5-2. In general, CoPCs for the terrestrial environment (i.e., antimony, barium, cadmium, lead, thallium, and zinc) were considered CoPCs for the land-based subsistence foods (caribou, ptarmigan, salmonberries, and sourdock). The CoPCs for the freshwater environment (lead and thallium) were considered CoPCs for fish. The data used to calculate EPCs for each of the subsistence foods are described below.

5.2.1.2.1 Data Used to Calculate Fish EPCs

Lead concentrations in fillets from adult Dolly Varden collected by DFG from the Wulik River from 1991 through 2003 were used in the risk assessment. These data are presented in Appendix G, Table G-30. The only other CoPC for the freshwater environment was thallium, which has not been analyzed in fish tissue. Fish tissue thallium concentrations were estimated as described in Section 5.2.1.2.6.1.

5.2.1.2.2 Data Used to Calculate Caribou EPCs

Data from ten adult caribou harvested in September 2002 by DFG personnel from locations along the DMTS were used in the risk assessment. Muscle, liver, and kidney tissue were analyzed for arsenic, cadmium, lead, and zinc concentrations. More detailed information on

---

[11] There are approximately 260 hectares in a square mile.

sampling locations and data analysis is presented in *Evaluation of Metals Concentrations in Caribou Tissues* (Exponent 2002e), which is included in Appendix H. Caribou tissue analytical data used in the risk assessment are presented in Appendix G, Table G-29. Arsenic data were not used in the risk assessment because arsenic was screened out and is not a site CoPC. The other CoPCs for the terrestrial environment are antimony, barium, and thallium, which were not analyzed in caribou tissue. Caribou tissue concentrations of these metals were estimated as described in Section 5.2.1.2.6.2.

5.2.1.2.3 Data Used to Calculate Ptarmigan EPCs

Five ptarmigan were collected from near the DMTS road in summer 2004, as described in the *Summary of Phase II Sampling Program for the DMTS Fugitive Dust Risk Assessment* (Appendix E) and shown in Figure 5-2. Muscle, liver, and kidney tissue were analyzed for antimony, barium, cadmium, lead, thallium, and zinc concentrations. Data from the three ptarmigan collected in the reference area were not used to calculate risks in the risk assessment. More detailed information on sampling locations and data analysis is presented in *Assessment of Metals in Ptarmigan Collected near the DMTS* (Exponent 2005b), which is included in Appendix H. Ptarmigan tissue analytical data used in the risk assessment are presented in Appendix G, Table G-27. Reference area ptarmigan data are presented in Appendix G, Table G-28.

5.2.1.2.4 Data Used to Calculate Salmonberry EPCs

As described in Section 5.2.1.1, for the purpose of the HHRA it was conservatively assumed that site soil concentrations are representative of conditions as far as 5 km downwind and 2 km upwind of the DMTS road and ambient air boundaries. Therefore, metals data from salmonberries collected from locations within the assumed site area were used in the risk assessment. The locations of these samples are shown in Figure 5-2; the data are presented in Appendix G, Table G-25, and include three sampling events: 1) samples collected in 2001 by E&E for DEC (E&E 2002); 2) samples collected in 2001 (Exponent 2002a); and 3) samples collected in 2004 (Exponent 2004c). Cadmium, lead, and zinc concentrations are available for the E&E samples. Concentrations of all six terrestrial CoPCs are available for the Exponent data. Data are available for both washed and unwashed samples, however to be conservative, only the unwashed sample data were used in the risk assessment. Although a number of the berry samples were collected within the DMTS ambient air boundary, where access is restricted and subsistence activities forbidden, these conservative data were still used in the risk assessment.

5.2.1.2.5 Data Used to Calculate Sourdock EPCs

Metals data from sourdock collected from locations within the assumed site area were used in the risk assessment. The locations of these samples are shown in Figure 5-2; the data are presented in Appendix G, Table G-26, and include two sampling events: 1) samples collected in 2001 by E&E for DEC (E&E 2002); and 2) samples collected in 2004 (Exponent 2004c). Cadmium, lead, and zinc concentrations are available for the E&E samples. Concentrations of all six terrestrial CoPCs are available for the Exponent data. Although data are available for both washed and unwashed samples, only the unwashed sample data were used in the risk assessment.

5.2.1.2.6 Estimation of CoPC Concentrations for Which Analytical Data Are Not Available

For fish and caribou, data were not available for all CoPCs. Specifically, there were no antimony, barium, or thallium data available for caribou, and no thallium data available for fish. For those CoPCs, concentrations were estimated as described below.

5.2.1.2.6.1 Fish

Lead and thallium were identified as CoPCs in the freshwater environment based on screening of surface water both as a drinking water source and as a source of fish for human consumption (see Sections 3.1 and 3.2). Although lead concentrations are available for fish tissue samples from the site, there are no data available for thallium. Therefore, an estimated EPC for thallium in fish tissue was derived using the relationship between thallium and lead concentrations in surface water (Table 5-3). Specifically, the mean thallium concentration in surface water was divided by the mean lead concentration in surface water. The resulting ratio of 0.17 was multiplied by the 95%UCL for lead in fish tissue to derive an estimate for thallium in fish tissue of 0.0026 mg/kg wet wt, which was applied as an EPC in the risk estimate.

This approach assumes that uptake of thallium in fish from water occurs at approximately the same rate as lead uptake. This assumption may over- or underestimate actual fish thallium concentrations. To evaluate this assumption, published bioconcentration factors (BCFs) for thallium and lead were compared. A BCF represents the relationship between the water concentration of a chemical and the fish tissue concentration of the chemical. The method used in this risk assessment assumes that the BCFs for lead and thallium are approximately the same. ATSDR (1999a) reports a median BCF value for lead in fish of 42. For thallium, ATSDR (1992c) reported a maximum BCF for bluegill of 34. Because these BCFs are similar, it is considered reasonable to use the ratio of thallium to lead in water to predict thallium concentrations in fish.

5.2.1.2.6.2 Caribou

Antimony, barium, cadmium, lead, thallium, and zinc were identified as CoPCs in the terrestrial environment based on screening of surface soils for incidental ingestion of soil under a residential use scenario. Although cadmium, lead, and zinc concentrations are available for caribou tissue samples (kidney, liver and muscle), no data are available for caribou antimony, barium, and thallium concentrations. Therefore, estimated EPCs for caribou CoPCs without data were derived using the relationships between the concentrations of those CoPCs in ptarmigan and cadmium, lead, and zinc in ptarmigan tissue (Table 5-4). Antimony was not detected in any of the ptarmigan tissue samples. Therefore, it was not included as a CoPC in caribou or ptarmigan. Thallium was not detected in ptarmigan breast tissue, and was detected in only one of five site ptarmigan liver samples at a concentration below that detected in a reference ptarmigan liver (0.0006 mg/kg versus 0.001 mg/kg, respectively). Thallium was detected in two of five site ptarmigan kidney samples, but one sample was at a concentration below that detected in a reference ptarmigan kidney (0.00049 mg/kg versus 0.0025 mg/kg, respectively) and the other was only slightly greater than the reference ptarmigan kidney (0.0037 mg/kg). Because thallium was not detected in the tissue comprising more than 90 percent of the food mass (i.e., muscle), was only detected in three organ samples, and when it was detected the concentration was near or below the level detected in reference animals, thallium was not included as a CoPC in caribou or ptarmigan.

A predicted caribou barium EPC was calculated using the most conservative estimate for each tissue (i.e., kidney, liver and muscle) and the relationships between barium and cadmium, lead, or zinc in ptarmigan. Specifically, in each type of ptarmigan tissue, the mean barium concentration was divided by the mean concentrations for cadmium, lead, and zinc. Of the resulting ratios, the maximum for each tissue was conservatively selected to derive the caribou barium EPCs. In all tissue types, the maximum ratio resulted from the relationship between barium and lead. For example, the mean ptarmigan muscle barium concentration (0.19 mg/kg) divided by the mean ptarmigan muscle lead concentration (0.025 mg/kg) resulted in a ratio of 7.67. This ratio was then multiplied by the caribou muscle lead 95%UCL (0.16 mg/kg) to predict a caribou muscle barium 95%UCL concentration of 1.2. The same procedure was used to predict caribou liver and kidney barium concentrations, as shown in Table 5-4. For all three tissue types, the ratio of barium to lead provided the highest ratio, and thus, the most conservative estimate of barium concentrations in caribou tissue.

This approach assumes that the ratio of barium to other metals in ptarmigan tissue will be similar to or greater than the ratio of barium to those metals in caribou tissue. This assumption may under- or overestimate the actual barium concentration in caribou tissue. There is a large degree of uncertainty in this method because of differences in metals uptake and metabolism between these animals, and because the ratios of barium to cadmium, lead, and zinc spanned more than two orders of magnitude. To address this uncertainty, the ratio that provided the most conservative (i.e., the highest) estimate of barium concentration in caribou tissue was used.

5.2.1.2.7 Estimation of Edible Tissue Weighted-Average Concentrations for Caribou and Ptarmigan

Subsistence food consumption rates are available for caribou and ptarmigan, but they are not broken down by tissue type. For example, there are no data available for the amount of caribou liver or ptarmigan kidney eaten. In order to match CoPC concentration data with consumption rate data, edible tissue weighted-average concentrations for caribou and ptarmigan were calculated.

A weighted-average concentration was calculated for edible caribou tissue using the percent weight contribution for each tissue type. As reported by ADPH (2001), both kidney and liver tissue contribute an estimated 2 percent of total caribou consumption, and muscle tissue contributes the remaining 96 percent. The value of 2 percent for caribou liver and kidney was estimated based on the percent weight of reindeer liver reported by Stimmelmayr (1994) and ADPH (2001). In that study, the tissue weighted-average EPCs for caribou were calculated using the following formula:

$$\text{Caribou EPC} = [\text{Kidney EPC} \times 0.02] + [\text{Liver EPC} \times 0.02] + [\text{Muscle EPC} \times 0.96]$$

A weighted-average concentration was calculated for ptarmigan edible tissue based on tissue weights reported by Kalas et al. (1995) for ptarmigan kidney, and by Remington and Braun (1988) for sage grouse liver and muscle tissue. No reports of ptarmigan liver and muscle weights were identified, so data from the most similar species available, the sage grouse, were used. It should be noted that the muscle weight represents only the breast and wing muscles. Thus, muscle tissue likely represents a larger portion of total ptarmigan consumption than assumed in this assessment, which would tend to result in a conservatively higher EPC calculation. As summarized in Table 5-5, edible tissue weights for kidney, liver, and muscle are estimated to represent 1, 9, and 91 percent of total edible tissue, respectively. Therefore the tissue-weighted-average EPCs for ptarmigan were calculated using the following formula:

$$\text{Ptarmigan EPC} = [\text{Kidney EPC} \times 0.01] + [\text{Liver EPC} \times 0.09] + [\text{Muscle EPC} \times 0.90]$$

The assumptions used regarding the relative proportion of total caribou consumption contributed by muscle, liver, and kidney are based on data reported by ADPH (2001). Based on that information, it is unknown whether leg and back muscle is included in the estimate of 96 percent of edible tissue as muscle. However, tissue weighted-average concentrations that do not include leg and back muscle provide a more conservative estimate of metals intake via caribou consumption because muscle tissue tends to have lower metals concentrations than liver or kidney tissue. Thus, the estimates used to calculate tissue weighted-average metals concentrations for caribou would be more likely to overestimate than underestimate total metals intake via caribou consumption.

For ptarmigan, tissue weighted-average concentrations were derived using only the weight of “breast” muscle (i.e., the pectoralis and supracorocoideus muscles). Similar to caribou, tissue weighted-average concentrations not including leg, wing, and back muscle provide a conservative estimate of metals intake from consumption of ptarmigan because muscle tissue tends to have lower metals concentrations than liver or kidney. In addition, ptarmigan comprise a very small portion of the subsistence diet so small changes in the ptarmigan consumption pathway exposure assumptions would have a negligible effect on overall risk calculations. In summary, the estimates used to calculate tissue weighted-average metals concentrations for both caribou and ptarmigan would be more likely to overestimate than underestimate total metals intake from caribou and ptarmigan consumption.

5.2.2 Subsistence Use

The subsistence use receptor scenario addresses exposures that could potentially occur as a result of subsistence food consumption, water ingestion, and the incidental soil and dust ingestion that might 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.

5.2.2.1 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 ALM; 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 ALM 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 was 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 ALM; therefore, use of the ALM for the subsistence use scenario is unnecessary. However, the ALM was applied to assess workers' cumulative exposures 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.[12] The IEUBK model (Windows Version 1.0) was 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 were EPA defaults (U.S. EPA 1994), with the exception of those input parameters for which site-specific information was available. Specifically, site-specific data for soil concentrations, gastrointestinal absorption for soil, drinking water concentration, and dietary intake were available and were used in the model. In addition, the soil lead EPC was multiplied by the fractional intake for the site of 0.09 to account for the fact that only a fraction of ingested soil would come from the site. The derivation of the fractional intake is described in Section 5.2.2.2, *Exposure Assumptions for Non-Lead CoPCs*. No information is available that would warrant modifying other default input, nor would any of the other parameters be expected, *a priori*, to differ for Northwestern Alaska. All input parameters used in the risk assessment are listed in Table 5-6, and the site-specific parameters are described below.

5.2.2.1.1 Soil Lead

The soil lead concentration input to the model was 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 chemical concentrations were related to fugitive dust. Thus, use of data only from the road and port industrial area provides a conservative estimate of chemical exposure from soil and dust.

[12] 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.

As shown in Table 5-1, the mean soil lead concentration in the port area is 1,255 mg/kg. In the road area, the mean soil lead concentration is 198 mg/kg. Using the methodology described in Section 5.2.1.1, area-weighted soil lead EPCs of 282 mg/kg and 726 mg/kg were calculated using the area-weighted and area-averaged approaches, respectively. As described above, these values were multiplied by the site fractional intake of 0.09 to account for the fact that only a fraction of ingested soil would come from the site. Thus, the soil lead concentrations used in the IEUBK model were 25 mg/kg and 65 mg/kg for the area-weighted and area-averaged approaches, respectively.

5.2.2.1.2 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 the default value of 30 percent. Likewise, the lead sulfate to which galena will eventually weather in the environment has also been identified by U.S. EPA (1999b) as having relatively low bioavailability. The Alaska Division of Public Health published data from a National Toxicology Program (NTP) study examining bioavailability of lead in Red Dog ore 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 5-7, Arnold and Middaugh (2001) reported relative bioavailability of lead in Red Dog ore-supplemented diets ranging from 13.6 percent to 27 percent associated with 100 ppm and 10 ppm, respectively, of lead in the diet. 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 the same amount of soluble lead acetate. The IEUBK model requires absolute bioavailability as an input (U.S. EPA 1996c). Absolute bioavailability is the fraction of ingested lead that enters the blood stream from the gastrointestinal tract and 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 to 13.5 percent. The average absolute bioavailability in the study was 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). Blood lead levels were even lower in residents of Kivalina and Noatak in a recent study conducted by Alaska Division of Public

Health (ADPH 2005). This suggests low bioavailability, low exposure, or both. The results of these blood lead studies are described in detail in Section 5.4.3.3.

There are two areas of uncertainty associated with the use of the NTP study results in the risk assessment. First, the NTP bioavailability study was conducted on Red Dog ore. After weathering, the lead in site soils may become more or less bioavailable. It should be noted, however, that many of the geochemical forms of lead that would most likely be formed from oxidation of lead sulfide in the environment (e.g., lead sulfites, lead sulfates, and lead oxides) are also considered by U.S. EPA (1999b) to have less than default bioavailability. Second, the NTP study used rats, whereas juvenile swine are the preferred animal model for development of site-specific bioavailability values (U.S. EPA 1999b). These issues are further discussed in the uncertainty assessment (Section 5.4.3), and addressed in the DMTS risk assessment by evaluating risks using both the IEUBK model default absolute bioavailability of 30 percent and the site-specific value of 9.7 percent.

5.2.2.1.3 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 (Table 5-1). The site arithmetic mean stream surface water lead concentration of 0.33 μg/L was used in the DMTS risk assessment.

5.2.2.1.4 Subsistence Food Lead

Model input for subsistence food lead intake was 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 (Table 5-8). Derivation of subsistence food consumption rates for the risk assessment is described in detail in Section 5.2.2.3. Those consumption rates were used to calculate food intake rates for the site, as described in Section 5.2.2.2. 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. Fish were identified in the CSM as representative of subsistence use in the area relevant to the freshwater environment. The average lead concentration for each subsistence food multiplied by the daily intake rate of that food gives an estimate of the daily intake of lead from each food source. The lead intake from all food sources was summed to give a total subsistence food lead intake from the DMTS site of 1.6 μg/day. This value was used in the IEUBK model. At the request of DEC, an alternative value for lead intake from all food sources of 3.4 μg/day was calculated using an alternative fractional intake for caribou of 0.2, as described in Section 5.2.2.2.3.

As described in Section 5.2.2.3, subsistence food intake rates were calculated assuming subsistence foods comprise the total diet, with no food intake from non-subsistence sources. Nevertheless, the IEUBK model was run assuming that lead intake from other food sources also continued because the default background dietary lead intake was included without subtracting out the portion of the diet comprised of site-derived subsistence foods. This background dietary lead is meant to include all dietary sources of lead. Therefore, adding the estimated dietary lead intake from site-related sources on top of default background dietary lead intake lends a conservative over-representation of lead intake to the exposure estimate, which is described further in the uncertainty assessment in Section 5.4.3.

5.2.2.2 Exposure Assumptions for Non-Lead CoPCs

Exposure to non-lead CoPCs was evaluated by combining estimates of media (soil, water, and food) intake with estimates of the chemicals concentrations in those media. Exposure to non-lead CoPCs in subsistence food and in soil and dust during subsistence hunting and gathering was evaluated in children and adults.

5.2.2.2.1 Soil

Soil exposure to non-lead chemicals was evaluated under the subsistence user scenario using standard EPA equations and RME assumptions (U.S. EPA 1989, 1991). Exposure concentrations for soil were calculated based on the lesser of the 95%UCL or the maximum detected concentration (Table 5-1). Exposure assumptions for adults are presented in Table 5-9 and for children in Table 5-10.

The estimated daily intake for each chemical from soil was calculated using the following equation:

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

where:

C_s	=	chemical concentration in soil (mg/kg-dry weight)	=	see Table 5-1
10^{-6}	=	conversion of mg soil to kg soil		
IR_s	=	soil ingestion rate (mg/day)	=	200 for children
			=	100 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)	=	0.09, see discussion below
BW	=	body weight (kg)	=	15 for children
			=	70 for adults
AT	=	averaging time (days)	=	2,190 for children
			=	10,950 for adults.

5.2.2.2.2 Water

Exposure concentrations for water were calculated based on the lesser of the 95%UCL or the maximum detected concentration for surface water from streams on the DMTS site (Table 5-1). Exposure assumptions for adults are presented in Table 5-9 and for children in Table 5-10.

The estimated daily intake of each chemical from water was calculated using the following equation:

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

where:

C_w =	chemical concentration in water (μg/L)	=	see Table 5-1
10^{-3} =	conversion of μg to mg		
IR_w =	water ingestion rate (L/day)	=	1 for children
		=	2 for adults
ED =	exposure duration (years)	=	6 for children
		=	30 for adults
EF =	exposure frequency (days/year)	=	365
FI =	fractional intake from site (unitless)	=	0.09, see discussion below
BW =	body weight (kg)	=	15 for children
		=	70 for adults
AT =	averaging time (days)	=	2,190 for children
		=	10,950 for adults.

5.2.2.2.3 Subsistence Food

Exposure to non-lead chemicals in subsistence foods was 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 was estimated using the following equation:

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

where:

C_f =	chemical concentration in food item (mg/kg-wet weight)	=	see Table 5-2
10^{-3} =	conversion of g food to kg food		
CR_f =	food item consumption rate (g/day)	=	see Table 5-11

ED =	exposure duration (years)	=	6 for children
		=	30 for adults
EF =	exposure frequency (days/year)	=	365
FI =	fractional intake from site (unitless)	=	0.09, see discussion below
BW =	body weight (kg)	=	15 for children
		=	70 for adults
AT =	averaging time (days)	=	2,190 for children
		=	10,950 for adults.

The estimated chemical intake from each food item was calculated individually so that risks could be expressed individually for each food item. The relative amount of subsistence food consumption related to gathering along the DMTS (versus from all subsistence use areas) was considered through application of a fractional intake term. Only a portion of soil and water ingestion would take place on the site. Similarly, only a portion of subsistence foods would be collected at the site, and for the animals used as subsistence foods, only a portion of their life would be spent at the site. The fractional intake term accounts for the fact that the area affected by the DMTS comprises only a portion of the total subsistence use area, and assumes that all areas within the subsistence use area are equally likely to be utilized. For wide ranging subsistence food animals, such as caribou and fish, the fractional intake term provides a conservative estimate of the portion of those animals' home ranges that the site covers. Fractional intake was derived by estimating the area of the site within the subsistence use area relative to the total subsistence use area. The fractional intake was calculated as follows:

$$\text{Fractional Intake} = \frac{\text{area of site within subsistence use area}}{\text{total subsistence use area}}$$

Available information on subsistence use areas for Kivalina and Noatak (Dames & Moore 1983a) was used in conjunction with conservative assumptions regarding the extent of the area represented by available concentration data. Figures 5-3 and 5-4 show subsistence use areas for Kivalina and Noatak, respectively. As described in Section 5.2.1.1, for the purpose of the HHRA it was conservatively assumed that site concentrations are representative of conditions as far as 5 km downwind and 2 km upwind of the DMTS. Thus, a band extending 5 km downwind and 2 km upwind of the road and the mine ambient air boundary is assumed to represent the "site."

For Kivalina, the total subsistence use area measures 549,352 hectares and the site area within that subsistence use area measures 48,617 hectares (Figure 5-3). The fractional intake based on the Kivalina subsistence use would therefore be 0.09 (i.e., 48,617/549,352). For Noatak, the eastern boundary of the subsistence use area is not reported by Dames & Moore (1983a), although it is clear that it continues past the limits of the area shown in their report. Using just the area reported by Dames & Moore (1983a), the total subsistence use area for Noatak measures >673,451 hectares and the site area within that subsistence area measures 69,725 hectares. The fractional intake based on these data would be 0.10. Because the measured Noatak subsistence use area was truncated at its eastern border, it is not as useful as the Kivalina data in calculating the fractional intake. In addition, if the full Noatak subsistence use area is only about 15 percent larger than the truncated area shown in Figure 5-4, it would give the same fractional intake as the Kivalina area. Based on the observed contour of the Noatak subsistence use area before reaching the truncation, it appears likely that true Noatak subsistence use area is more than 15 percent larger than the truncated area. Thus, the fractional intake of 0.09 calculated from the Kivalina subsistence use area information was used in the risk assessment.

For berries and sourdock, only data from samples collected at the site were used in the risk assessment. Ptarmigan may have a home range that extends outside the assumed DMTS site area. But a ptarmigan home range is much smaller than that of a caribou or fish, so it is more likely that an animal collected on the site would have spent a significant portion of its life on the site. The metals concentrations in berries, sourdock, and ptarmigan tissue collected at the site were considered to be representative of metals concentrations in all berries, sourdock, and ptarmigan tissue collected for subsistence food from the assumed DMTS site area. Furthermore, for the risk assessment it was conservatively assumed that all the metals in those foods were related to DMTS fugitive dusts and not from background metals. Therefore, it is appropriately conservative to use the site fractional intake of 0.09 (described previously) to estimate the fraction of metals intake from berries, sourdock, and ptarmigan consumption that is site-related.

Caribou and fish (specifically, Dolly Varden) have much larger home ranges than ptarmigan, and in fact, much larger than the total subsistence use areas of the residents of Kivalina and Noatak. Because of this, caribou and fish tissue metals concentrations used in the risk assessment already integrate these animals' exposure over the total subsistence use areas, and beyond. Therefore, it may not be appropriate to apply a fractional intake that represents the fraction of subsistence caribou and fish that are collected from the site. Also, metals concentrations in tissue of these animals reflect metals uptake from background sources outside the assumed DMTS site to a much greater extent than metals uptake from the site. The aim in the risk assessment is to estimate exposure to site-related metals. Therefore, while it may not be appropriate to apply a fractional intake that represents the fraction of subsistence caribou and fish that are collected from the site, it is appropriate to apply a fractional intake that represents the fraction of metals in caribou and fish tissue that is site-related. The latter is based on the fraction of those animals' total home range covered by the site (as a surrogate for the fraction of the animals time spent at the site). Because both caribou and fish have home ranges extending beyond the total subsistence use areas for Kivalina and Noatak, that fraction would be smaller than the fractional intake of 0.09 calculated for the other subsistence foods. However, the fractional intake of 0.09 was also used for caribou and fish tissue to provide a more health-protective evaluation.

An additional set of risk estimates was calculated using an alternative caribou fractional intake of 0.2 because of the uncertainty surrounding the amount of impact site metals might have on caribou tissue metals concentrations, and because of the unique role of caribou in the diet and culture of people from the region. At the request of DEC, this alternative value was calculated using the area reported to have cadmium levels elevated above background by Hasselbach et. al. (2005) as the site harvest area.

5.2.2.3 Review of Existing Subsistence Food Consumption Rate 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 rate 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 5-11. 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 5-11, 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 et al. (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-hour 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 5-12; 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 illustrate the degree to which the CPDB database overestimates intake. In addition, the data provided by CPDB include only subsistence foods, whereas the Nobmann et al. (1992) data 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, conservatively using the higher food intake rates reported for males (and as summarized in Tables 5-11 and 5-12):

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 average of the estimates for Kivalina and Noatak is 830 g/day (Table 5-11).

2. Food consumption estimates in the CPDB database are given in g/day. 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/g) 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 5-12):

   – 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). This differs slightly from the 2,750 kcal daily caloric intake reported by Nobmann et al. (1992), likely because of the standard rounding used for the specific energy content of protein, fat, and carbohydrates. The values calculated in Table 5-12 are used solely for the purpose of calculating the average caloric density of the diet.

   – 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 5-11).

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 5-11).

5. Caloric intake-weighted consumption rates for adults were derived by multiplying the CPDB-derived consumption rates by the caloric intake weighting factor (Table 5-11).

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, consumption rates were used only for the categories identified in the work plan for evaluation in the risk assessment. These consumption rates were 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.

In summary, the following subsistence food consumption rates were calculated:

- Caribou: 168 g/day for adults and 84 g/day for children, representing the consumption rate of all land mammals.

- Ptarmigan: 2.0 g/day for adults and 1.0 g/day for children, representing the consumption rate of ptarmigan, which was the only game bird for which consumption was reported.

- Fish: 124 g/day for adults and 62 g/day for children, representing the consumption rate of all non-salmon fish.

- Salmonberries: 8.4 g/day for adults and 4.2 g/day for children, representing the consumption rate of all berries.

- Sourdock: 1.3 g/day for adults and 0.7 g/day for children, representing the consumption rate for all plants, greens, and mushrooms.

The consumption rates for marine-based subsistence foods, such as marine mammals and marine invertebrates, were not used in the risk assessment because the marine environment was screened out. The consumption rate for salmon was not used because it is an anadromous fish that would spend very little of its life in the streams and creeks near the DMTS. In fact, the fish that comprises most of the non-salmon fish intake, Dolly Varden, is also anadromous. The consumption rate for Dolly Varden (see Table 5-11) was nevertheless included in the risk assessment. This provides a more conservative evaluation because it tends to overestimate the consumption of fish that have been exposed to site metals to any appreciable extent.

5.2.3 Combined Occupational and Subsistence Use

The occupational and subsistence use receptor scenario addresses exposures to future hypothetical DMTS workers that could potentially occur both at work on or near the DMTS study area and through subsistence activities occurring outside of work. Work-related exposures could occur through incidental soil and dust ingestion and potential subsistence exposures include 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 described first for lead and then for the remaining CoPCs.

5.2.3.1 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 ALM, which is recommended for adults and older children, is applied here to evaluate lead uptake. The EPA ALM 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 associated with a given soil concentration). Although EPA indicates that 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. Although the DMTS has signage in place to limit exposure, the risk assessment assumes that exposure will occur. While it may be unlikely that a woman in the 8th or 9th month of pregnancy would be working along the DMTS, it is possible that a woman could be working along the DMTS in the early stages of pregnancy or for the months prior to pregnancy.

The ALM 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. The model applies a biokinetic slope factor to exposure estimates in order to derive an estimate of blood lead concentrations related to exposure levels. Ingestion exposure is the primary pathway evaluated in the model. 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 2.3.3.1 of the CSM, most inhaled dust reaches only the upper respiratory tract, where it is carried into the esophagus and ultimately ingested. Exposure from inhaled dust is, in fact, 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)	=	see Table 5-1
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_w}	=	fractional intake of soil from site while working (unitless)	=	0.67
FI_{s_s}	=	fractional intake of soil from site during subsistence activities (unitless)	=	0.03
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:

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

where:

IR_f	=	daily lead intake from subsistence foods (g/day)	=	see Table 5-13
EF_f	=	exposure frequency for subsistence food (days/year)	=	182.5
AF_f	=	absorption fraction from food (unitless)	=	0.2

All other parameters are as above. 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 representing variability of blood lead in the population 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	=	population geometric standard deviation (unitless)	=	2.11
1.645	=	95th percentile value for the 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 (unitless)	=	0.9.

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

5.2.3.1.1 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. Although the Alaska Division of Public Health (ADPH) conducted blood lead studies in Kivalina and Noatak in 1991 (ADPH 2001) and 2004 (ADPH 2004), there were too few study participants to provide an adequate measure of baseline blood lead to confidently use in the ALM. The 1991 study evaluated a larger number of people, but the data are not representative of current conditions. As noted in Section 5.4.3.3, blood lead levels have decreased since 1991 in Kivalina and Noatak, as they have throughout the U.S. Therefore, the reported mean baseline blood lead of 1.53 μg/dL reported by U.S. EPA (2002a) for all ethnic groups combined from throughout the U.S. was used as input for the ALM in the risk assessment.

5.2.3.1.2 Soil and Dust Ingestion Rate

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 ALM guidance (U.S. EPA 1996c). An alternative method of separating soil lead and dust lead ingestion would be necessary only if there were more than one separate source with a different concentration and/or characteristic (e.g., if the contribution from lead-based paint is being assessed). At the request of DEC (2004b), a soil ingestion rate of 100 mg/day was used in the draft risk assessment for the worker/subsistence user for the time apportioned to subsistence activities and a rate of 50 mg/day was used for the portion of time a person would be at work. An adult soil ingestion rate of 50 mg/day is supported by both DEC (2002) and U.S. EPA (1996c, 1997b) guidance. U.S. EPA (1996c) states that a soil ingestion rate of 50 mg/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.” Inputs to the ALM should be central tendency estimates, rather than upper end estimates. Accordingly, U.S. EPA (1996c) recommends a default soil ingestion rate of 50 mg/day for use in the model. Nevertheless, in accordance with a later request by DEC, a soil ingestion rate of 100 mg/day is used for both the subsistence and worker portions of the subsistence/worker scenario. The impact on the model results of using the default soil ingestion rate of 50 mg/day rather than 100 mg/day is evaluated in the uncertainty assessment. The impact on the model results of using the default soil ingestion rate of 50 mg/day rather than 100 mg/day is evaluated in the uncertainty assessment.

5.2.3.1.3 Exposure Frequency to Soil

EPA recommends an exposure frequency for workers of 219 days per year for the ALM (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 was used as input for the ALM.

5.2.3.1.4 Gastrointestinal Absorption Fraction of Lead from Soil

The ALM default for soil lead bioavailability is 12 percent (U.S. EPA 1996c). As discussed in Section 5.2.2.1 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, as demonstrated by the lower default value for soil lead bioavailability for adults relative to children. Therefore, lead risks were modeled using both the model default and site-specific bioavailability.

As with the site-specific soil lead bioavailability for children discussed in Section 5.2.2.1, a site-specific soil lead bioavailability can also be calculated using the data from Arnold and Middaugh (2001) (Table 5-7). The default absolute bioavailability of soluble lead in the IEUBK model is 50 percent (U.S. EPA 1994), whereas the default bioavailability for soluble lead in the adult model is 20 percent (U.S. EPA 1996c), because of the reduced absorption of lead observed in adults relative to children. The site-specific data can be applied to the ALM 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) (i.e., $0.20 \times 0.194 = 0.039 = 3.9$ percent). Thus, in the DMTS risk assessment; risks were evaluated using both the ALM default soil bioavailability of 12 percent and the site-specific value of 3.9 percent.

5.2.3.1.5 Fractional Intake of Soil from Site

The fractional intake of soil from the site used in the risk assessment for the ALM (Table 5-13) is time-weighted to account for the difference in fractional intake that occurs during the two-thirds of the time that a person is on work rotation versus the one-third of the time they are off work. Specifically, it was assumed that during the two-thirds of the time that a person is working, 100 percent of soil ingestion occurs at the site. Thus, the weighted fractional intake of soil from the site while working (FI_{s_w}) is 1.0 (i.e., 100 percent) multiplied by two-thirds, or 0.67. The weighted fractional intake of soil from the site while engaged in subsistence activities

(FI_{s_s}) is the fractional intake described in the subsistence user scenario (i.e., 0.09) multiplied by one-third, which gives 0.03.

5.2.3.1.6 Lead Intake from Subsistence Foods from Site

IR_f was calculated in the same way as the child subsistence food lead intake rate used in the IEUBK model, which was described in Section 5.2.2.1. The average lead concentration for each subsistence food was multiplied by the daily intake rate of that food to give an estimate of the daily intake of lead from each food source. The lead intake from all food sources was summed to give a total subsistence food lead intake from the site for adults of 1.6 and 3.4 μg/day for the site fractional intake and alternative caribou fractional intake scenarios, respectively (Tables 5-13 and 5-14). This value was used in the ALM. The IR_f accounts for the fractional intake from the site because the daily food consumption rates were calculated using the fractional intake of 0.09.

5.2.3.1.7 Exposure Frequency for 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 was 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. Although workers are not allowed to subsistence hunt or gather while on a work shift, it is possible that some subsistence foods are brought from home. The amount of subsistence food consumption that occurs while on a work shift is unknown, but it would be expected to be low because all meals are provided in the Red Dog cafeteria. However, because some subsistence food consumption may occur, for the DMTS risk assessment it was 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, the EF_f used in the DMTS risk assessment was 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).

5.2.3.1.8 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 ALM guidance. Therefore, an assumption analogous to that recommended in the IEUBK model guidance was used in the DMTS risk assessment. The default absolute absorption assumed for soluble lead in the ALM is 0.2 (i.e., 20 percent) (U.S. EPA 1996c). Thus, an absorption fraction for lead in foods of 0.2 was used in the ALM for the DMTS risk assessment. Evidence from studies in humans conducted by U.S. EPA Region 3 scientists in collaboration with other researchers indicates that this fraction is conservative. Based on lead absorption in adult volunteers under fasting and non-fasting conditions, a time-weighted average absorption of lead from soil is about 0.06 for someone consuming three meals per day and 0.0825 for someone consuming two meals per day (Maddaloni et al. 1998). The default fraction of 0.2 would be conservative because a pregnant woman (the receptor for the ALM) is not likely to be in a fasting condition for any length of time.

5.2.3.1.9 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 inadequate data available for either the specific region or ethnic group relevant to this risk assessment. Although ADPH conducted blood lead studies in Kivalina and Noatak in 1991 (ADPH 2001) and 2004 (ADPH 2004), there were too few study participants to provide an adequate measure of variability in the blood lead levels to confidently use in the ALM. Therefore, the combined GSD of 2.11 for all ethnic groups in all regions of the U.S. was used as input for the ALM in the risk assessment.

5.2.3.2 Non-Lead CoPCs

Exposure to non-lead CoPCs (Table 5-15) was evaluated by combining estimates of media (soil, water, and food) intake with estimates of the chemical concentrations in those media (i.e., the EPCs calculated as described in Section 5.2.1).

5.2.3.2.1 Soil

Soil exposure to non-lead CoPCs for the combined worker and subsistence hunter and gatherer scenario was evaluated using standard EPA equations and RME assumptions (U.S. EPA 1989, 1991).

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

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

All input assumptions for soil exposure in this scenario are the same as described for the adult subsistence user scenario (in Section 5.2.2.2), with the exception of the soil ingestion rate while at work (IR_{s_w}), the fractional intake of soil from the site while at work (FI_{s_w}) and during subsistence activities (FI_{s_s}), and the exposure duration (ED), which have been modified to account for the cumulative amount of exposures workers who also consume subsistence foods may experience. As described above for lead exposure, an FI_{s_w} of 0.67 was applied to account for the fact that two-thirds of the time (i.e., while at work) essentially all soil exposure is assumed to take place at the site (i.e., during work that is assumed to occur solely on and near the DMTS), and an FI_{s_s} of 0.03 was applied to account for the fact that one-third of the time (while off work rotation), only 9 percent of soil intake is assumed to take place at the site (i.e., during subsistence gathering activities near the road). Also, as described above for lead, a soil ingestion rate of 100 mg/day was used for both the portion of time a person is at work and off work. The EPA default ED of 25 years is shorter than the ED used in the adult subsistence user scenario of 30 years in concordance with DEC guidance for a commercial/industrial exposure scenario (DEC 2002). As a practical matter, the specific ED used does not matter for calculation of intake of non-cancer chemicals; mathematically, whatever value is used for ED cancels out with AT because the AT is equal to ED.

5.2.3.2.2 Water

Exposure to non-lead CoPCs in stream surface water for the combined worker and subsistence hunter and gatherer scenario was evaluated using standard EPA equations and RME assumptions (U.S. EPA 1989, 1991).

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

The intake equation and all input assumptions for water exposure in this scenario are the same as described for the adult subsistence user scenario (in Section 5.2.2.2), with the exception of ED and the fractional intake (FI_{ww}). The EPA default exposure duration of 25 years is shorter than the exposure duration of 30 years used in the adult subsistence user scenario in concordance with DEC guidance for a commercial/industrial exposure scenario (DEC 2002). The adjusted fractional intake accounts for the fact that stream water intake would be different during subsistence hunting and gathering activities than while at work. An FI_{ww} of 0.045 was used in the risk assessment (Table 5-15). The rationale for this value is similar to that for food intake, and is discussed below, under Worker's Consumption of Subsistence Food.

5.2.3.2.3 Workers' Consumption of Subsistence Food

Exposure estimates for non-lead chemicals in subsistence foods in the combined worker and subsistence user scenario used 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_{wf}):

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

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 was 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., $FI_{w\_off} = FI = 0.09$, or 0.2 for caribou using the alternative caribou fractional intake). 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 was 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., $FI_{w\_on} = 0.25FI = 0.0225$). Thus, FI_{wf} was 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:

$$\begin{aligned} FI_{wf} &= (0.33 \times FI_{w\_off}) + (0.67 \times FI_{w\_on}) \\ &= (0.33 \times 0.09) + (0.67 \times 0.0225) \\ &= (0.0297) + (0.0151) \\ &= 0.045 \end{aligned}$$

Using the same calculations but with the alternative caribou FI_{w_off} of 0.2, FI_{wf} for caribou would be calculated as 0.1.

Although intake of stream water while on a work shift would be considered even less likely to occur than subsistence food consumption, it could not be ruled out. Therefore, the same assumptions were made for water intake as for subsistence food intake and a fractional intake of water for workers (FI_{ww}) of 0.045 was calculated. Both the food and the water fractional intake are expected to overestimate exposure for most individuals, but are included to provide a health-protective estimate of exposures and risks.

5.3 Toxicity Assessment

In the toxicity assessment, the hazards associated with CoPCs 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 antimony, barium, cadmium, thallium, and zinc are published in EPA's Integrated Risk Information System and are available online (U.S. EPA 2005). EPA has not developed an RfD for lead, but rather evaluates lead toxicity in reference to blood lead levels, as described in Section 5.2.2.1. None of the site CoPCs is classified by EPA as a carcinogen for the exposure routes relevant to this assessment. The toxicity criteria used in the risk assessment are summarized in Table 5-16. The following subsections provide the toxicity assessments for the CoPCs evaluated in the HHRA: antimony, barium, cadmium, lead, thallium, and zinc.

5.3.1 Antimony

Antimony occurs naturally in soil at low concentrations, with a reported average concentration in soil in the Western U.S. of 0.76 mg/kg (Dragun and Chiasson 1991). Data for Alaska were not identified in the sources reviewed, but it is notable that antimony can be locally elevated in ore-bearing areas. Data are limited on the forms of antimony in the environment. Antimony released into waterways is typically in particulate forms. Data are limited regarding concentrations of antimony in marine or freshwater resources, but where available, concentrations appear to be less than 5 μg/L. For example, ATSDR (1992a) identified a dissolved antimony concentration of 0.332 μg/L for the Yukon River. The primary source of antimony exposure for most people is the diet. The U.S. Food and Drug Administration estimates that the average concentration of antimony in the diet is 9.3 μg/kg with a daily dietary intake of 4.6 μg/day for someone with a 3,076 kcal daily caloric intake (ATSDR 1992a).

U.S. EPA (2005) has derived an RfD for oral exposure to antimony of 0.0004 mg/kg-day based on data from a chronic oral exposure study in rats. Schroeder et al. (1970) dosed 50 male and

50 female rats with antimony tartrate in water in a chronic study. Treated rats had a shortened lifespan: male rats survived 106 and females 107 fewer days than did control rats that did not receive antimony. Treated animals were also observed to have lower heart weights, lower blood glucose levels, and alterations in cholesterol levels relative to controls. The study was not adequate to derive a NOAEL because only one dose was administered. Therefore, EPA based the RfD on the lowest-observed-adverse-effect level (LOAEL) of 0.35 mg/kg/day, which was the dose level reported by the authors. The RfD was derived through application of an uncertainty factor of 1,000. This uncertainty factor included a factor of 10 to account for interspecies variability, a factor of 10 to protect sensitive individuals, and an additional factor of 10 because the effect level was a LOAEL and not a NOAEL.

U.S. EPA (2005) indicated that adverse effects on the heart have been reported in workers exposed to high concentrations in industrial settings and indicated that an RfD for this endpoint would be approximately 0.003 mg/kg-day. Thus, the existing RfD of 0.0004 mg/kg-day would be protective of this endpoint. EPA does not consider antimony to be a carcinogen based on available evidence.

5.3.2 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 1992b). Barium levels in seawater range from 2 to 63 mg/L (Bowen 1979). In all surface water, concentrations range from 2 to 380 mg/L (ATSDR 1992b). 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 1992b). 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 at 5 to 150 mg/kg, and in various other vegetables at 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 (2005) has established an RfD for barium of 0.2 mg/kg-day. The RfD was derived based on a chronic exposure study conducted by the NTP in both rats and mice (NTP 1994). Animals were exposed to barium chloride dihydrate in their drinking water at concentrations of 0, 125, 500, 1,000, 2,000, and 4,000 mg/L. In this study the kidney was identified as the most sensitive organ to the effects of long-term exposure barium. Specifically, evidence of nephropathy (i.e., adverse effects in the kidney) in mice was reported primarily in the renal tubules, and these effects were determined to be distinct from the age-related pathology typically seen in rodents. Effects were also noted for rats, but there was no detectable difference between rats exposed to barium and those in the control group. Thus, results based on kidney effects in mice were used to derive the RfD.

Although mild to moderate nephropathy was observed in a few animals in the intermediate dose groups, significant effects were seen only at the highest dose. However, rather than use a NOAEL as the basis for deriving the RfD, EPA used a benchmark dose approach whereby a mathematical model is fit to the response data for all doses used in the study to predict a dose at which only a minimal response (5 percent response rate) would be expected. The 95 percent lower confidence limit of the "benchmark dose" associated with the 5 percent response rate ($BMDL_{05}$) was used as the point of departure, rather than a NOAEL, to derive the RfD. The $BMDL_{05}$ of 63 mg barium/kg-day was subsequently divided by an uncertainty factor of 300 (10-fold to account lack of knowledge about the relative sensitivity of humans relative to mice, 10-fold to account for lack of knowledge about differing sensitivities between people, including children, and 3-fold for database deficiencies, including lack of information regarding the potential for reproductive and developmental effects). The resulting RfD of 0.2 mg/kg-day was used in the DMTS risk assessment.

U.S. EPA (2005) 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 (2005) was used in the DMTS risk assessment.

5.3.3 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.

EPA has established two oral RfDs for cadmium: one for food and one for water (U.S. EPA 2005). 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 2005). 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, 2005). 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.[13] In commenting on the Thun et al. (1985) study, EPA noted that:

> "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, 2005).

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 as low as 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 as high as 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}).

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

5.3.4 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 $\mu g/m^3$ in rural settings to an average of 0.04 $\mu g/m^3$ in urban settings. The NAAQS for lead is 1.5 $\mu g/m^3$. Lead is also present in foods. As reported in ATSDR (1999a), data from the U.S. Food and Drug Administration'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 2002a). 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 2002a). 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 2002b). 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 lead exposure prior to birth resulted in no effect 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 2005). 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 2005). U.S. EPA (2005) considers data from the studies to be inadequate to determine the carcinogenic potential of lead. All available studies in humans 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 noncarcinogenic effects.

5.3.5 Thallium

Thallium occurs naturally in soil. Concentrations in the earth's crust are estimated to be between 0.3 mg/kg and 0.7 mg/kg (ATSDR 1992c). Thallium is infrequently detected in drinking water in the absence of any known source. A survey of tap water from 3,834 homes in the U.S. found detectable thallium in 0.68 percent of samples at an average concentration of 0.89 μg/L. Seawater concentrations of thallium are reported to range from 0.01 to 14 μg/L (Sharma et al. 1986, as cited in ATSDR 1992c). Thallium is found in trace amounts in most foods and this is the most common source of exposure for most people, with an estimated daily intake of 5 μg/day for a typical 70 kg person (ATSDR 1992c).

The EPA oral RfD for thallium of 8×10^{-5} mg/kg-day was derived from a 90-day (subchronic) study in rats dosed with thallium sulfate in water (U.S. EPA 1986, as cited in U.S. EPA 2005; U.S. EPA 2005). The RfD was derived from the NOAEL of 0.25 mg/kg-day determined in the study. No differences between the control groups and groups receiving thallium sulfate were observed in body weights, body weight gains, food consumption, or absolute and relative organ weights. Some dose-related changes were observed in some blood chemistry parameters: increased SGOT, LDH, and sodium levels, and decreased blood sugar levels. The only grossly observed finding at necropsy thought to be treatment-related was alopecia (i.e., hair loss), especially in female rats. However, microscopic evaluations did not reveal any histopathologic alterations. Moreover, alopecia was observed in both treated and nontreated rats (likely a result of grooming behavior [U.S. EPA 1986]), which reduces the likelihood that this was a treatment-related effect. EPA identified the 0.25 mg/kg-day dose level for thallium sulfate to be a NOAEL in this study. EPA derived the RfD by applying an uncertainty factor of 3,000 to this

NOAEL. EPA notes that the of 3,000 includes factors of 10 to extrapolate from subchronic to chronic data, 10 for intraspecies variability, 10 to account for interspecies variability, and 3 to account for lack of reproductive and chronic toxicity data.

5.3.6 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 2005).

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 2005; 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).

5.4 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 are estimated assuming long-term exposure to contaminants detected in site media. There were no site CoPCs classified as carcinogens by EPA for the exposure routes relevant to this assessment.

Risks from lead exposure are evaluated by estimating blood lead levels using EPA's IEUBK model for children (Table 5-17), which was described above in Section 5.2.2.1, and EPA's ALM for evaluating risks to the fetuses of pregnant adult workers (Tables 5-18 and 5-19). All lead modeling results are summarized in Table 5-20. 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.

The risk characterization methods described in DEC and EPA guidance were applied to calculate hazard indices for CoPCs other than lead. 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.

Hazard quotients were calculated for each CoPC (other than lead) for each of the primary exposure pathways identified in the refined CSM. The potential for cumulative effects from multiple CoPCs within an exposure pathway was evaluated by summing the hazard quotients for all CoPCs in that pathway. The potential for cumulative effects across all exposure pathways was evaluated by summing the hazard indices for all pathways for each receptor.

Potential risks associated with exposure to non-lead metals are presented by pathway in Tables 5-21 through 5-47, and summarized across all pathways in Tables 5-48 through 5-51.

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.

In the remainder of this section, risk estimates are first presented for the subsistence user scenario and then for the combined worker/subsistence user scenario. In addition, the major uncertainties associated with the risk assessment are discussed and, to the extent possible, the degree to which they might over- or underestimate risk is evaluated.

5.4.1 Risk Estimates for the Subsistence Use Scenario

Risk estimates for the subsistence use scenario are described in the following subsections for the lead and non-lead CoPCs.

5.4.1.1 Risk Estimates for Lead

To evaluate risks from exposure to lead in the subsistence use scenario, child blood lead levels were modeled using EPA's IEUBK model and through application of the default and site-specific input values summarized in Table 5-6. Results from the IEUBK model are presented in Table 5-17. Lead risks were estimated using both the default lead bioavailability of 30 percent and the site-specific value of 9.7 percent, both the site fractional intake of 0.09 and the alternative caribou fractional intake of 0.2, and both the area-weighted and area-averaged soil lead concentrations. Results of the model for the different scenarios ranged from a predicted geometric mean blood lead of 1.0 μg/dL, with less than 0.0005 percent probability of exceeding the 10 μg/dL, to 1.9 μg/dL with 0.023 percent chance of exceeding 10 μg/dL. In all cases, the risks were well below EPA's target of 10 μg/dL geometric mean blood lead with less than a 5 percent probability of exceeding that target.

5.4.1.2 Risk Estimates for Non-Lead CoPCs

To evaluate risks from exposure to non-lead CoPCs in the subsistence use scenario, hazard quotients were calculated for each CoPC in each exposure pathway (i.e., soil ingestion, water ingestion, and consumption of each food item). These risk estimates are shown in Tables 5-21 through 5-38. In addition, combined risks for all CoPCs in each exposure pathway, and for all pathways combined, are summarized in Tables 5-48 through 5-51.

For adults, the combined hazard index for all CoPCs and all pathways was 0.1 for site-specific fractional intake (Tables 5-48 and 5-49) and 0.2 assuming an alternative caribou fractional intake (Tables 5-50 and 5-51). The largest contributor to the adult subsistence use risk was caribou consumption, with a hazard index of 0.07 assuming site-specific fractional intake, or 76 percent of the total risk estimate. The CoPC contributing most significantly to the risk estimate for adult subsistence use caribou consumption was cadmium (hazard quotient=0.05), followed by zinc (hazard quotient=0.03) (Table 5-27).

For children, the combined hazard index for all CoPCs and all pathways was 0.3 for site-specific fractional intake (Tables 5-48 and 5-49) and 0.5 assuming an alternative caribou fractional intake (Tables 5-50 and 5-51). The largest contributor to the child subsistence use risk was caribou consumption, with a hazard index of 0.2 assuming site-specific fractional intake, or 68 percent of the total risk estimate. The CoPC contributing most significantly to the risk estimate for child subsistence caribou consumption was cadmium (hazard quotient=0.1), followed by zinc (hazard quotient=0.06) (Table 5-28). Soil ingestion contributed 14 percent of the total risk estimate for children using area-weighted concentrations, with a hazard index of 0.04. The CoPCs contributing most significantly to the risk estimate for child soil ingestion were barium and antimony, both with hazard quotients of 0.01 assuming area-weighted soil concentrations) (Table 5-23).

5.4.2 Risk Estimates for the Combined Worker/Subsistence Use Scenario

Risk estimates for the combined worker/subsistence use scenario are described in the following subsections for the lead and non-lead CoPCs.

5.4.2.1 Risk Estimates for Lead

To evaluate risks from exposure to lead in the combined worker/subsistence use scenario, blood lead levels were modeled for the fetus of a pregnant woman using EPA's ALM and the input values summarized in Table 5-13. As described in Section 5.2.3.1, lead is less bioavailable in adults than in children. Thus, the ALM default bioavailability is 12 percent, and the site-specific value is 3.9 percent. For the risk assessment, lead risks were estimated using both the default lead bioavailability of 12 percent and the site-specific value of 3.9 percent, both the site fractional intake of 0.09 and the alternative caribou fractional intake of 0.2, and both the area-weighted and area-averaged soil lead concentrations. Results of the model for the different scenarios ranged from a predicted geometric mean blood lead in the fetus of a woman exposed at the site of 1.6 μg/dL, with a 0.7 percent probability of exceeding the 10 μg/dL, to 2.7 μg/dL with 4.0 percent chance of exceeding 10 μg/dL. In all cases, the risks were well below EPA's target of 10 μg/dL blood lead with less than a 5 percent probability of exceeding that target assuming either site-specific fractional intake (Table 5-18) or the alternative caribou fractional intake (Table 5-19). In fact, these results approach the lower limit of the model's ability to predict risk. Even if there were no exposure from site soil or subsistence food lead, the model would still predict a geometric mean blood lead level of 1.4 μg/dL, with a 0.4 percent probability of exceeding 10 μg/dL. This is because the model assumes a certain baseline level of blood lead.

5.4.2.2 Risk Estimates for Non-Lead CoPCs

To evaluate risks from exposure to non-lead CoPCs in the combined worker/subsistence use scenario, hazard quotients were calculated for each CoPC in each exposure pathway (i.e., soil ingestion, water ingestion, and consumption of each food item). These risk estimates are shown in Tables 5-39 through 5-47. In addition, combined risks for all CoPCs in each exposure pathway, and for all pathways combined, are summarized in Tables 5-48 through 5-51.

In the combined worker/subsistence use scenario, the combined hazard index for all CoPCs and all pathways ranged from 0.08 assuming site-specific fractional intake and area-weighted soil concentrations (Tables 5-48 and 5-49) to 0.1 assuming an alternative caribou fractional intake and area-averaged soil concentrations (Tables 5-50 and 5-51). The largest contributor to this risk estimate was caribou consumption, with a hazard index of 0.04 (49 percent of the total hazard index), followed by soil ingestion, with a hazard index of 0.03 (38 percent of the total) assuming site-specific fractional intake. The CoPCs contributing most significantly to the risk estimate for soil ingestion were antimony and barium, both with hazard quotients of 0.01 assuming area-weighted soil concentrations (Table 5-39). The CoPC contributing most significantly to the risk estimate for caribou consumption was cadmium (hazard quotient=0.02), followed by zinc (hazard quotient=0.01) (Table 5-42).

5.4.3 Uncertainty Assessment

Because risk characterization serves as a bridge between risk assessment and risk management, it is important that major assumptions, scientific judgments, and estimates of uncertainties be described in the assessment. Risk assessment methods are designed to be conservative to address the uncertainties associated with each step in the risk assessment process. Thus, “true” site risks are likely to be less than risks estimated using standard risk assessment methods.

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. The primary uncertainties in the DMTS HHRA are discussed in the following sections.

5.4.3.1 Reference Area Selection

There are two general ways in which a potential depositional influence on the reference areas could affect the risk assessment: 1) The validity of the reference comparison used in the CoPC screening procedures, and 2) the conclusions in the risk assessment that are based on reference area comparisons. The following discussion reviews the process used to select the reference area, the role of the reference area data in CoPC screening, and the impact of reference area data on risk characterization and the quantitative risk estimates on which it is based.

5.4.3.1.1 Reference Area Selection Process

The reference area selection process is summarized below. Additional details are provided in the ERA uncertainty assessment (Section 6.6).

Terrestrial reference areas were selected after review of existing studies and data, with a focus on factors such as prevailing wind directions, bedrock geology, topography and physiography (including slope, aspect, and water features such as streams and tundra ponds), and plant and animal communities. Possible reference areas were considered to the east, north, west, and south of the mine and DMTS. The prevailing wind originates from the east, between the northeast and southeast quadrants; thus, the most significant dust deposition has occurred to the north and west of the DMTS road and mine. As a result, areas to the north and west were not preferred areas for establishing the terrestrial reference area. Areas to the east were eliminated because the topography is more mountainous than most of the DMTS area. Thus, the focus was on selecting an area to the south of the mine and DMTS road. However, selecting an area too far south would have put the reference area into the Noatak valley, where the plant community includes trees and would not be as good for comparison with plant communities at the site. Therefore, the terrestrial reference area was targeted for placement somewhere within several miles south of the DMTS. Within that band south of the DMTS, the selected area was to be in a geologic area known to be relatively free of lead/zinc base metal mineralization. The selected area also needed to contain a variety of topographic conditions (elevations, slopes, and aspects), streams and ponds, and plant communities, providing the opportunity to sample environments similar to those along the length of the DMTS road. Based on these criteria, the Evaingiknuk Creek drainage was selected as the best choice. This basin met the most criteria, and had low base metal mineralization compared with other possible reference locations that were considered to the south of the DMTS.

Subsequent to the selection of the Evaingiknuk Creek drainage as the terrestrial reference area, sampling was conducted in two phases. The first phase included sampling of moss, which, when included with the overall moss database (including the NPS data, Ford and Hasselbach 2001, Hasselbach 2003, pers. comm., Hasselbach et al. 2005) and plotted together, provided a clearer perspective on overall patterns of deposition in the areas surrounding the DMTS and mine (Figure 1-9). Prior to the first phase of sampling, no moss data were available in that area. The mean lead concentration for the three moss samples in the reference area was 8.0 mg/kg. Tundra soil was also sampled in the reference area, and the lead concentration ranged from 2.9 to 23.3 mg/kg, with a mean of 8.9 mg/kg, very similar to the mean moss lead concentration. In the area beyond approximately 16 miles north of the DMTS, where there was no apparent trend in the NPS moss concentration data, the mean lead concentration in moss was 8.5 mg/kg, or 6.4 if one outlier duplicate sample is excluded (Dixon's outlier test was used to confirm that the 38.6 ppm lead result is a statistical outlier at the 0.05 level [$0.02 < P < 0.05$], among the samples with similar concentrations greater than 16 miles north of the DMTS). Concentrations in the reference area and the area beyond 16 miles north of the DMTS appear to be similar. In the southern extent of the CAKR, beyond 12 to 13 miles south of the DMTS, the NPS moss lead concentrations averaged 2.0 mg/kg. It should also be noted that the area surrounding the Red Dog district is more mineralized than the southern part of CAKR. If there were dust depositional influence in the reference area, or the northern extent of the data collection area, it would appear to be very limited.

5.4.3.1.2 CoPC Screening

Selection of CoPCs for the HHRA was generally a two-step process: First, site chemical concentrations were statistically compared to reference concentrations. Second, site concentrations were compared to human health-protective risk-based screening levels (DEC 2003a) derived using conservative residential use assumptions, 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 medium, those chemicals that both exceeded their risk-based screening level *and* were significantly different from reference concentrations were retained as human health CoPCs. Thus, comparisons between site and reference data were particularly important for CoPC selection when chemical concentrations at the site were above risk-based screening levels. The following chemicals had site concentrations that exceeded risk-based screening levels, but were eliminated from further consideration in the baseline risk assessment because site concentrations were not statistically significantly greater than reference concentrations:

- Soil: Aluminum, arsenic, iron, and manganese
- Stream water: Aluminum, barium, and iron
- Lagoon water: Arsenic and manganese
- Marine water: Arsenic and manganese
- Marine sediment: None (site concentrations were below the SQS).

Overall, the HHRA results indicated little or no risk from the CoPCs carried through the assessment. If risks are low for the site CoPCs, which are the metals that drive risks at the site, then risks would be even lower for metals that were screened out. During development of the risk management plan (discussed in Section 8.1), the risk assessment results can be used to prioritize future actions such as additional data collection or monitoring. If there are future changes in site concentrations of metals that were screened out by comparison with reference areas, and those changes are related to fugitive dust deposition, the changes will be detected by concomitant changes in the concentrations of CoPCs that are included in future monitoring programs.

5.4.3.1.3 Risk Characterization

Reference comparisons were not used as primary evidence for evaluating human health risks in the baseline risk assessment. However, tissue CoPC concentrations in ptarmigan from the terrestrial reference area provided supporting evidence for eliminating thallium as a CoPC in ptarmigan and caribou. Specifically, thallium was not detected in ptarmigan breast tissue, and was detected in only one of five site ptarmigan liver samples at a concentration below that detected in a reference ptarmigan liver (0.0006 mg/kg and 0.001 mg/kg, respectively). Thallium was detected in two of five site ptarmigan kidney samples, but one sample was at a concentration below that detected in a reference ptarmigan kidney (0.00049 mg/kg and 0.0025 mg/kg, respectively) and the other was only slightly greater than the reference ptarmigan kidney (0.0037 mg/kg). Because thallium was not detected in the tissue comprising more than 90 percent of the food mass (i.e., muscle), was only detected in three organ samples, and when it was detected the concentration was near or below the level detected in reference animals, thallium was not included as a CoPC in caribou or ptarmigan. The low frequency of detection of thallium in tissue samples from the site also supports a conclusion of low human health risk from thallium exposure without direct comparisons to reference data, and therefore the reference comparison was not pivotal to the overall risk characterization for this chemical.

5.4.3.2 Adult Lead Model Inputs

5.4.3.2.1 Soil Ingestion Rate

Data on adult soil ingestion are limited, and no quantitative information on soil ingestion during subsistence activities is available. Thus, the soil ingestion rate during subsistence activities is an area of uncertainty. As requested by DEC during work plan comment resolution, a soil ingestion rate during subsistence activities of 100 mg/day was used as an input to the ALM.

Subsequently, as part of comment resolution following submittal of the draft risk assessment, DEC requested that an adult soil ingestion rate of 100 mg/day be applied during work time as well. U.S. EPA (1996c) recommends 50 mg/day as central estimate and 100 mg/day as a high-end estimate, based on the best available data. In addition, U.S. EPA (1996c) further notes that 100 mg/day is used to represent agricultural exposure scenarios in EPA risk assessments. For the ALM, a value of 100 mg/day likely overestimates actual exposure because: 1) the ALM is designed to use average values as input assumptions, not upper end estimates; 2) EPA guidance indicates that an ingestion rate of 50 mg/day adequately addresses incidental soil and dust ingestion (U.S. EPA 1996c); and 3) DEC (2002) recommends an adult soil ingestion rate of 50 mg/day to calculate cleanup levels for commercial/industrial settings. In fact, if a soil ingestion rate of 50 mg/day were used instead of 100 mg/day for the adult worker/subsistence use scenario, and all other exposure assumptions remained the same, the results for the ALM would not change because the low fractional intake for soil ingestion during subsistence activities minimizes the sensitivity of the model to this parameter.

5.4.3.2.2 Soil Lead EPC

Based on site use characteristics, it was assumed that exposure occurs randomly throughout the site. Thus, area-weighted EPCs were calculated that represent integrated exposure over the entire site. During the time that people are at work, their activities would not be expected to keep them in one location over days and months, and during the time they are off work and engaged in subsistence activities, they would, in fact, be expected to spend little or no time in the vicinity of the port, where the highest lead concentrations are located. Nevertheless, in order to evaluate the potential impact of higher soil lead concentrations at the port on the results, risks were estimated using the average soil lead concentration in the port area, 1,255 mg/kg.

Using a soil lead exposure concentration of 1,255 mg/kg in conjunction with the model default bioavailability of 12 percent for the adult worker/subsistence use scenario, the predicted geometric mean blood lead level was 2.6 μg/dL, with a 3.4 percent chance of exceeding 10 μg/dL. This is still below EPA's target. When used in conjunction with the site-specific bioavailability of 3.9 percent, the predicted geometric mean blood lead level dropped to 1.8 μg/dL, with a 1.2 percent chance of exceeding 10 μg/dL, well below the target.

Effect of changing soil EPC on the results from the adult lead model for the adult worker/subsistence use scenario

	Assuming Bioavailability of 12 percent		Assuming Bioavailability of 3.9 percent	
	GeoMean Blood Lead (μg/dL)	Percent Chance of Exceeding Target	GeoMean Blood Lead (μg/dL)	Percent Chance of Exceeding Target
Soil EPC = 282 mg/kg	1.7	0.9	1.6	0.7
Soil EPC = 1,255 mg/kg	2.6	3.4	1.8	1.2

5.4.3.2.3 GSD

A GSD of 2.11 was used as an input to the ALM. The GSD represents variability in blood lead levels over the entire population studied. In this case, it represents the variability for all women in the U.S., ages 17–45. Generally speaking, variability would tend to decrease as the population in which the measurements were made becomes more homogeneous (i.e., more alike in race, age, exposure patterns, etc.). The adult female population in Kivalina or Noatak is far more homogeneous, both in demographics and in exposure patterns, than the entire adult female population of the U.S. With increased homogeneity, there is decreased variability, and thus, the GSD would be expected to decrease. In fact, using the port site average soil concentration of 1,255 mg/kg and the model default bioavailability of 12 percent, the model would predict the site to meet EPA's targets even if variability were greater, as reflected in a GSD as high as 2.28.

5.4.3.3 Child Lead Model Inputs

5.4.3.3.1 Site-Specific Bioavailability

There are two areas of uncertainty associated with the use of the NTP study results in the risk assessment. First, the NTP bioavailability study was conducted on Red Dog ore, not surface soil lead. When the ore concentrate particles, primarily galena, are exposed to air and water in the environment, over time the surfaces of these particles could become more oxidized. Increased oxidation could, in turn, increase solubility, which could be associated with increased bioavailability (Brown et. al. 1999). With environmental weathering, the lead in site soils may become more or less bioavailable in the environment. While there are no data available on the bioavailability of soil lead along the DMTS corridor, USGS (2003) has reported on the mineralogy of lead in Red Dog ore concentrate, port soil, Ikalukrok creek alluvium, and colluvial samples from deposits in the area. Scanning electron microscopy shows that galena particles in port soil exhibit morphology similar to ore galena particles: well-developed cubic cleavage with smooth faces. This is in contrast to galena particles from stream alluvium, which are rounded from physical/mechanical processes, and from colluvial samples, which are etched and rounded. It is noteworthy that neither the soil nor the alluvial galena particles are etched, indicating less oxidation than in colluvial samples, which could be related to a lack of acidic conditions. In any case, it should be noted that many of the geochemical forms of lead that would most likely be formed from oxidation of lead sulfide in the environment (e.g., lead sulfites, lead sulfates, and lead oxides) are also considered by U.S. EPA (1999b) to have less than default bioavailability. Thus, the approach used in the risk assessment of estimating risks based on both the IEUBK model default absolute bioavailability of 30 percent and the site-specific value of 9.7 percent should adequately address this area of uncertainty.

The second area of uncertainty associated with the NTP study is the animal model used. Juvenile swine are the preferred animal model for development of site-specific bioavailability values (U.S. EPA 1999b). However, the NTP study used rats. This area of uncertainty is somewhat mitigated by the fact that the results are based on relative, not absolute bioavailability. Specifically, the data resulting from the NTP study provide an estimate of the bioavailability of concentrate ore lead *relative* to soluble lead acetate. The resulting relative bioavailability is then applied to the EPA default value for *absolute* bioavailability of soluble lead acetate. Although there may be differences in absolute lead bioavailability between animal species related to differences in their respective digestive systems, the differences in relative bioavailability of lead from two sources should be less. This is because much of lead bioavailability is related to its ability to go into solution (i.e., solubility); the higher the solubility, the greater the bioavailability. This is the basis of the *in vitro* bioaccessibility test used to estimate bioavailability. Lead bioaccessibility testing measures the potential of lead from a test source to go into solution, relative to lead acetate, under acidic and basic conditions designed to mimic the gastrointestinal system. The results of this test provide a surrogate for relative bioavailability. In a similar way, the NTP study should provide a reasonable estimate of the solubility, and thus the bioavailability, of lead from Red Dog ore *relative to* lead acetate.

5.4.3.3.2 Soil Lead EPC

Based on site use characteristics, it was assumed that exposure occurs randomly throughout the site. Thus, area-weighted EPCs were calculated that represent integrated exposure over the entire site. In fact, if a child were to be present during subsistence activities at the site, he or she would be less likely to be exposed to the higher concentrations present at the port. Because of this, the area-weighted lead EPC would tend to overestimate risks. Nevertheless, because of this area of uncertainty, risks were estimated both with and without area weighting. Without area weighting, EPCs will be skewed toward concentrations at the port, where the majority of soil samples have been collected. This provides an additional level of health-protectiveness in the risk assessment.

5.4.3.3.3 Soil Ingestion Fractional Intake

Soil lead EPCs used in the IEUBK model were adjusted for use in the DMTS risk assessment to account for the fact that only a fraction, if any, of a child’s daily soil ingestion would occur at or near the DMTS. Specifically, the soil lead concentrations calculated for the site with and without area weighting were multiplied by the site fractional intake of 0.09.

Typically, soil ingestion is one of the exposure pathways that tend to drive risk estimates when evaluating childhood lead risks. Therefore, in order to evaluate the effects of this site-specific modification (i.e., use of fractional intake = 0.09) on the results, the IEUBK model was run using varying assumptions about fractional intake to determine what fractional intake, if any, would result in a 5 percent chance of exceeding the target blood lead level of 10 μg/dL. As shown in the table below, even at a fractional intake of 1.0 (i.e., all soil ingestion occurs at the site), exceedance of the EPA target blood lead level is not predicted to occur using the area-weighted soil lead concentration, even assuming both the high end soil lead bioavailability of 30 percent and the high end caribou consumption fractional intake. Using the area-averaged soil concentration, fractional intake would need to be greater than 50 percent to predict an exceedance of the EPA target blood lead level assuming a soil lead bioavailability of 30 percent. The model predicted target blood lead level cannot be achieved assuming the site-specific bioavailability of 9.7 percent. Thus, predicted risks are still low under most scenarios even without applying a fractional intake to soil ingestion, and are exceeded under the area-averaged soil concentration only when assuming greater than 50 percent fractional intake in conjunction with high soil lead bioavailability.

Soil ingestion fractional intake necessary to exceed target blood lead level in child lead model

	Assuming Bioavailability of 9.7 Percent	Assuming Bioavailability of 30 Percent
Area-Weighted Soil Concentration		
Site caribou fractional intake	>1.0	>1.0
Alternative caribou fractional intake	>1.0	>1.0
Area-Averaged Soil Concentration		
Site caribou fractional intake	>1.0	0.56
Alternative caribou fractional intake	>1.0	0.51

5.4.3.3.4 Dietary Lead

Dietary lead intake was estimated by combining lead concentrations in subsistence foods with the intake rates for those foods and the fractional intake from the site. The estimated lead intakes from all subsistence food items were then summed. The resulting value, 1.6 μg/day, was considered to represent potential lead intake from subsistence foods obtained from the site. The updated model default dietary lead intake (3.16, 2.60, 2.87, 2.74, 2.61, 2.74, 2.99 μg/day for 0–1, 1–2, 2–3, 3–4, 4–5, 5–6, and 6–7 year olds, respectively) was also included to account for other dietary sources of lead.

In fact, because the model default dietary intake of lead is meant to include all sources of dietary lead from a complete diet, any additional dietary input to the model overestimates the amount of food eaten and the actual dietary lead intake. Separate caribou, salmonberry, sourdock, and ptarmigan evaluations indicate that metals levels in subsistence foods in the area are similar to subsistence foods elsewhere (Appendix H, *Subsistence Foods Data Evaluations*, and summarized in Section 5.4.3.7 of this document). Thus, including the additional dietary lead intake of 1.6 μg/day from subsistence foods taken at or near the site double counts part of the dietary lead input to the model, which contributes to a more health-protective evaluation.

Lead concentrations in fillets from adult Dolly Varden collected by DFG from the Wulik River from 1991 through 2003 were used in the risk assessment to estimate the fish lead EPC. Other fish organs may also be consumed, but tissue-weighted concentrations were not calculated for fish as they were for caribou and ptarmigan (described in Section 5.2.1.2.7). Although muscle tissue comprises most of the edible portion of the fish, portions of the fish not included could contribute to lead exposure. Of particular interest would be bone, where lead may accumulate. There is uncertainty regarding the concentrations of lead in fish bones, the amount of bone consumed by people, and the associated contribution to estimated risks. This uncertainty is partly addressed by the fact that subsistence dietary lead is included in the IEUBK child lead model in addition to the default dietary lead intake included in the model. Even with this overestimate of total dietary lead, predicted risks were very low. Inclusion of other portions of the fish would be expected to have little to no impact on the risk estimates because 1) tissues other than muscle comprise a relatively small percentage of total fish consumption, 2) lead concentrations do not differ significantly between whole body fish (which includes bones) and muscle or other tissues (e.g., liver and kidney) of Dolly Varden collected in the Wulik by

DFG (Scannell 2005), and 3) intake of lead from fish is less than 4 percent of total estimated dietary lead intake (Table 5-8).

5.4.3.4 Discussion of ADPH Blood Lead Surveys

ADPH conducted blood lead surveys in Kivalina and Noatak in 1990 (ADPH 2001) and 2004 (ADPH 2005). In the most recent study, blood from 10 individuals from Kivalina and 48 from Noatak was analyzed for lead and cadmium levels. Two of the 10 individuals from Kivalina and two of the 48 from Noatak were in the 6–18 age range. All others were over 18 years of age. There were no young children (age 0–6 years of age) included.

None of the 58 individuals had a blood lead level exceeding 10 μg/dL. Among the Kivalina participants, the geometric mean blood lead among individuals over 18 years of age was 1.1 μg/dL, with individual blood lead levels ranging from less than 1 up to 7 μg/dL. Among Noatak residents, the geometric mean blood lead level among individuals over 18 years of age was 1.7 μg/dL, with individual blood lead levels also ranging from less than 1 up to 7 μg/dL. It is noteworthy that the geometric mean values in both Kivalina and Noatak are less than or equal to the geometric mean for adult women estimated by the ALM for this risk assessment. As shown in Tables 5-18 and 5-19, the ALM predicted geometric mean blood lead levels ranging from 1.6 μg/dL to 2.7 μg/dL, depending on assumed soil lead concentration, soil lead bioavailability, and fractional intake of caribou from the site. Blood cadmium levels were similarly low.

The biomonitoring survey provides important public health information directly to the individuals who participated in the study, and is useful as supporting information for the risk assessment process. However, the study cannot, nor was it designed to, provide direct input to the lead exposure models used in the risk assessment. There were no study participants in the 0- to 6-year-old range, the age group evaluated in the IEUBK child lead model and the target population for the subsistence use scenario. Of the participants in the 18-and-older group, the report does not segregate data by gender and specific age group. Women of child-bearing age (approximately 18 to 45 years of age) are the population evaluated in the ALM and are the target population for the worker/subsistence use scenario.

Even if the individual data were obtained so blood lead levels could be analyzed by gender and age, several other study design issues preclude its use in population level models such as those used in the risk assessment, including: 1) individuals were included in the study if they volunteered rather than being randomly selected to participate, which is important because 2) relatively few individuals participated, particularly from Kivalina, and 3) even if the participant selection process were random, the small number of participants would provide a more uncertain estimate of the geometric mean blood lead and, especially, the GSD in blood lead. For example, the values recommended by EPA and used in the risk assessment are based on a random survey among more than 5,000 individuals (U.S. EPA 2002a). EPA notes that the "perceived gains in specificity achieved…" by using data that have been split up in a way that presumably more closely matches site characteristics "…may be offset by increased uncertainty caused by using less of the available survey data." (U.S. EPA 2002a). This statement was made in reference to stratifying the national database by more than one factor (e.g., region and race), but applies in this case as well because EPA was referring to sample sizes in hundreds, which is much larger than the 58 individuals in the 2004 ADPH study, presumably only a portion of whom were women of child-bearing age.

Although interpretation of the results of the 2004 blood lead survey from a population level standpoint is limited by the small numbers of participants and the lack of data for small children (0–6 years old), the survey data are consistent with the following observations:

1. The assumptions used in the risk assessment are unlikely to underestimate exposure. Risk assessments, in general, use conservative exposure assumptions to ensure that even sensitive individuals and those with higher than normal exposure are protected. In this case, the blood lead data indicate blood lead levels that are less than those predicted.
2. Lead at the site has a lower bioavailability than the default level used in the ALM and IEUBK child lead model. The outputs from the ALM using site-specific bioavailability were closer to the measured blood lead in the communities.
3. Blood lead levels in Kivalina and Noatak appear to be dropping. The geometric mean blood lead levels were lower in 2004 than in 1990 for both Kivalina and Noatak. In addition, 32 of the 33 individuals who participated in both the 1990 and 2004 surveys had lower blood lead in 2004.

ADPH (2004) concluded that "all blood lead and cadmium results are below levels that are of public health concern" and that the "…results provide additional evidence that the villages of Noatak and Kivalina are not being exposed to lead or cadmium from mining operations."

5.4.3.5 Estimated Fish and Caribou CoPCs

The lack of analytical data for some CoPCs in fish (thallium) and caribou (antimony, barium, and thallium) adds a level of uncertainty into the risk assessment. Rather than proceed without quantitative estimates of risk from these CoPCs, available data from other media were used to estimate concentrations of these CoPCs in fish and caribou.

For fish, the relationship between thallium and lead in water was used to estimate thallium concentrations in fish. This assumes that uptake of thallium in fish occurs at approximately the same rate as lead uptake. This assumption may over- or underestimate actual fish thallium concentrations. To evaluate this assumption, we compared published BCFs for thallium and lead, as described in Section 5.2.1.2.6. Because the BCFs were similar, uptake was assumed to be similar. Depending on the study design and fish species used, BCFs calculated for a given chemical may vary by one or more orders of magnitude. But given the low predicted thallium risk estimates from fish consumption (0.001 for adults and 0.05 for children), this comparison suggests that use of the relative concentrations of thallium and lead in water to predict fish tissue thallium concentrations is reasonable.

The relationship between barium and other CoPCs in ptarmigan tissue was used to estimate barium concentrations in caribou tissue. This method assumes that the ratios of barium to other metals in ptarmigan tissue are similar or greater than the same ratios in caribou tissue, and thus, use of these ratios is a health-protective means to evaluate barium in caribou tissue. The assumption that the relationship between metals is consistent across species is an area of uncertainty in the risk assessment. While acknowledging this uncertainty, an effort was made to reduce the possibility of underestimating risk by using the metal that produced the highest ratio in ptarmigan tissue to predict the barium concentration in the corresponding caribou tissue.

5.4.3.6 Exposure Frequency for Soil Ingestion

An exposure frequency of 200 days per year was applied to estimate exposure via the soil ingestion pathway for all metals for adults, and for non-lead metals for children. The site fits the arctic zone criteria of snow coverage or frozen ground for at least 165 days per year, as indicated in DEC (2002) guidance. U.S. EPA (2003d) indicates that soil ingestion during the winter may be greatly reduced because of snow cover and frozen ground. Although EPA notes that soil ingestion can continue at a lower level in the winter months through tracking outdoor soil inside and through contact with indoor dust in the home, they are referring to situations where outdoor soil is still intermittently not snow-covered and not frozen during winter months, which is not the case in the arctic zone of Alaska. Also, dust inside Kivalina and Noatak residences would have little to no impact from the site because of the distance from the DMTS. The majority of soil ingestion occurs through hand to mouth contact. During snow coverage there would be no direct contact with outdoor soil. When the ground is frozen, soil would be physically less available for ingestion because it would not adhere to skin in the same way as dry, thawed soil. Likewise, dust that has settled onto the snow would be frozen and would not adhere to the skin in the same way as dry, thawed soil. In addition, people’s skin, including their hands, would be covered during much of the year, limiting hand to mouth contact.

Based on DEC (2002) and U.S. EPA (2003d) guidance, our understanding of the site, and the dynamics of the soil ingestion pathway, we believe the recommended arctic zone exposure frequency of 200 days per year is appropriate for the site. The IEUBK model for child lead exposure was applied assuming a more conservative model default exposure frequency of 365 days per year. The minimal impact on risk estimates that would occur as a result of using the more accurate exposure frequency does not warrant the complicated adjustment necessary to incorporate this less conservative modification into the IEUBK model.

5.4.3.7 Fractional Intake

Fractional intake from the site is an area of uncertainty. Fractional intake is intended to account for the fraction of total media exposure (soil, water, berries, sourdock, and ptarmigan) that occurs at the site. For stationary subsistence foods (i.e., berry and sourdock) and foods with a small home range (i.e., ptarmigan) the fractional intake represents the fraction of that food type collected from the site relative to all areas where it is collected. It is true that harvesting can occur only where the food item is available, and not evenly throughout the subsistence harvest area. However, in the absence of data to the contrary, it is a reasonable assumption that a person would be equally likely to harvest a given food on a similarly sized area off the site and on the site. As an example, berries do not grow evenly throughout the site. However, the proportion of the “site” harvest area covered by berries can reasonably be assumed to be similar to the proportion of the non-site harvest area covered by berries. And if a person is equally likely to harvest from each of the berry harvesting areas, a fractional intake based just on berry harvesting areas would be the same as the fractional intake that was calculated based on the entire harvest use area. A person may, in fact, be more likely to use a berry harvesting area nearer to home, which would be offsite, than one onsite that is farther away (and off limits). Thus, it is reasonably likely that the fractional intake, as calculated, overestimates fractional intake from the site.

For subsistence food animals with large home ranges (caribou and fish), fractional intake is intended to account for the fraction of the animal's life that is spent at the site, and thus the fraction of metal content in the animal that is theoretically attributable to the site. As with the plant foods and ptarmigan, it is based on the area of the site relative to the total area of subsistence harvest. For caribou and fish, the metals concentrations in those animals used in the risk assessment already integrate the animal's exposure over their entire home range. But only a fraction of the metals detected in these animals would have been derived from site exposure. Given that there appears to be no significant difference in metals concentrations in site caribou relative to caribou from elsewhere in Alaska (Appendix H), it can be inferred that site caribou do not appear to have been exposed to greater amounts of metals at the site than elsewhere in their home range. Thus, the fraction of metals detected in those caribou that could be attributed to site exposure can be estimated by the fraction of time spent at the site relative to elsewhere in their home range, which can in turn be estimated by the fraction of the area of the site relative to their entire home range. In fact, the home ranges for both caribou and fish are far larger than the subsistence harvest areas for Kivalina or Noatak. Thus, the fractional intake used in the risk assessment likely greatly overestimates the fraction of metals in these animals that is attributable to the site. In addition, as noted above and detailed in Appendix H, the results of the caribou metals evaluation suggest that metals concentrations in caribou harvested at the site are not elevated relative to background. If that were indeed the case, any risk estimate based on caribou metals concentrations, regardless of the fractional intake applied, would be an overestimate of site-related risks.

While it is difficult to quantify the exact fractional intake, it can be estimated using knowledge of use patterns. For the DMTS risk assessment, three primary sources of information were used to estimate fractional intake: 1) previously published information on the extent of subsistence use areas for Kivalina and for Noatak (Dames & Moore 1983a,b); 2) knowledge of the nature and extent of metals concentrations around the DMTS; and 3) information about standard work schedules at the Red Dog mine.

The estimated fractional intakes used in the risk assessment (0.09 in the subsistence use scenarios; 0.67 and 0.03 (while off work) for soil ingestion and 0.045 for food/water consumption in the worker/subsistence use scenario) may over- or underestimate the actual fractional intake from the site. This issue is partly addressed by inclusion of risk estimates using an alternative caribou fractional intake of 0.2, as described in Section 5.2.2.2.3. To further address this uncertainty, the effect of altering the fractional intake on the estimated risks from exposure to non-lead metals was evaluated.

For the child subsistence use scenario, a cumulative hazard index of 1.0 is estimated only when the assumed fractional intake is 0.36 (i.e., 36 percent of all soil, water, and food consumption was from the site). If a fractional intake of 1.0 is assumed (i.e., that 100 percent of all soil, water, and food consumption was from the site), the resulting cumulative hazard index is 2.9. While this hazard index exceeds the target of 1.0, it is still within the degree of uncertainty inherent in the RfDs used to calculate risks. In addition, risks from individual CoPCs are not typically considered cumulative and summed unless the target organ and mechanism of action on which the RfD is based are the same. Only two CoPCs (i.e., barium and cadmium) have RfDs based on effects in the same target organ (the kidney). In reality, the fractional intake from the site would never be 1.0 for a child, and the fractional intake of 0.09 used in the risk assessment likely significantly overestimates an actual child's contact with the site.

For both the adult subsistence use and the combined worker subsistence use scenarios, a cumulative hazard index of 1.0 was estimated only when the assumed fractional intake was 0.95 (i.e., 95 percent of all soil, water and food consumption was from the site). If a fractional intake of 1.0 is assumed, the resulting cumulative hazard index is 1.1. Again, this is within the degree of uncertainty inherent in RfD derivation, and no individual CoPC exposure would result in a cumulative hazard index exceeding 1.0, even with a fractional intake of 1.0. Although an adult may come into contact with the site to a greater degree than a child, an actual adult would still never attain 95 percent of their soil, water, and food from the site. Furthermore, site restrictions do not allow subsistence harvesting on the site at all, and the DMTS road does not increase access and exposure to the site, because the road is designated strictly for industrial use. Public use of the road is not permitted. Access control practices for mine, DMTS port and DMTS road facilities are defined and regulated by the air quality permits for the mine and DMTS port (No. 289TVP01 Revision 1, 290TVP01, and AQ0289MSS01). Additionally, the DMTS port facility public access control plan (Teck Cominco 2004k) is specifically referenced and required by the DMTS port air permits and ADNR Tideland Lease Amendment No. ADL 412501. The only time subsistence users would be on the road is to cross it at one of the designated crossing points. Crossing of the road at other points is not permitted. Crossing of the port facility is permitted along the designated beach corridor, and large warning signs are posted at either end of the beach crossing. In addition, security of the port is also regulated under 33 CFR Subchapter H (homeland security requirements for maritime operations).

Effect of changing fractional intake on estimated risks for non-lead metals

Scenario	Cumulative Hazard Index Associated with: Site-Specific Fractional Intakes	Cumulative Hazard Index Associated with: Fractional Intake=1.0	Fractional Intake Associated with Cumulative Hazard Index=1.0
Child subsistence use	0.3	2.9	0.36
Adult subsistence use	0.1	1.1	0.95
Worker/subsistence use	0.08	1.1	0.95

5.4.3.8 Cumulative Risk Estimates

According to EPA guidance, cumulative risk assessment evaluates risks from multiple chemicals through all exposure pathways. A cumulative risk assessment should consider the combined health effects of a group of chemicals with a common mechanism of action, defined as two or more chemicals "that produce an adverse effect(s) to human health by the same, or essentially the same, sequence of major biochemical events. The underlying basis of the toxicity is the same, or essentially the same, for each chemical" (U.S. EPA 1998). Thus, risks from multiple chemicals should only be summed if those chemicals operate through the same mechanism. DEC (2002) guidance provides the same direction, indicating that cumulative risk should be addressed by calculating a hazard index, where "HI is the summation of all of the HQs for all pathways and exposure routes that affect the same target organ or system endpoint."

Of the CoPCs evaluated in the DMTS risk assessment, only the RfDs of cadmium and barium are based on effects in the same organ system (i.e., the kidney, see Table 5-16). Thus, summing the hazard quotients for all CoPCs to derive a total hazard index for each receptor overestimates actual site risks.

5.4.3.9 Chemicals Lacking Adequate Toxicological Information

Nine chemicals (bismuth, calcium, chloride, gallium, germanium, gold, silicon, sulfate, and sulfur) were excluded from the quantitative risk assessment. As discussed in Section 3 of this document, these constituents (with the exception of calcium) are not on EPA's target analyte list or DEC's list of hazardous substances for which cleanup levels are provided. The DEC risk assessment procedures manual (DEC 2000) explains that these lists were developed using the Pareto principle, which 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."

The general basis for EPA and DEC's exclusion of these chemicals is, in part, the lack of adequate toxicological information. There are no relevant toxicity criteria for these constituents and, because of the lack toxicological data, toxicity criteria cannot be derived as part of this assessment. Therefore, quantitative risk estimates based on exposure to these constituents is not possible.

Exclusion of bismuth, calcium, chloride, gallium, germanium, gold, silicon, sulfate, and sulfur introduces additional uncertainty into the risk assessment. However, the impact of this uncertainty on the overall results of the risk assessment is minimized by the fact that these constituents are generally not considered to be environmental hazards. In addition, 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.

5.4.3.10 Discussion of Previous Subsistence Foods Investigations

Three investigations have previously been conducted to evaluate whether subsistence foods present in the vicinity of the DMTS might be affected by metals from DMTS fugitive dust. These evaluations include the following: 1) caribou sampled in 1996 and 2002 (Exponent 2002e; 2) berries and sourdock sampled in 2001 and 2004 (Exponent 2004d); and 3) ptarmigan sampled in 2004 (Exponent 2005b). These investigations are each discussed briefly here and the methods and findings are described in detail in Appendix H.

5.4.3.10.1 Caribou

Caribou were sampled in 1996 and in 2002 at locations near the Red Dog mine and from other areas of Northern Alaska (Exponent 2002e [see Appendix H], Garry et al. 2004). The data used for the risk assessment were from caribou harvested after over-wintering near the DMTS. Thus, they were harvested during a period of time when any metals exposure related to the site would have still been reflected in their soft tissues. Caribou muscle, liver, and kidney tissue were analyzed for lead, zinc, cadmium, and arsenic. Samples from caribou collected in the vicinity of the mine were compared with those from other areas in Northern Alaska and with metals concentrations identified in Canadian caribou and Scandinavian reindeer.

By comparison with Northern Alaska caribou metals concentrations, there were no apparent significant elevations in tissue metals concentrations in the 2002 Red Dog caribou samples. None of the metals were consistently higher or lower in all tissues of the Red Dog caribou relative to caribou or reindeer from Canada, Scandinavia, or elsewhere in Northern Alaska. Although several potential differences were noted between the 2002 Red Dog data and the comparison groups, the biological relevance and/or importance for human health is unclear. For example, although lead is one of the two primary constituents of the concentrates produced at the mine, muscle lead concentrations in area caribou do not appear to differ from those found in the U.S. meat supply (ATSDR 1999a).

Results from the risk assessment indicate that the metals concentrations detected in caribou at the site were not associated with elevated human health risks. However, the results of the comparison with metals concentrations in caribou from other areas of Alaska and the world suggests that even these low risk estimates are more related to background exposure than to site-related metals. Thus, the results of the caribou study are supportive of the conclusion that the fractional intake assumption used in the risk assessment is conservative, and health protective.

Despite evidence that caribou metals concentrations were similar to background, those concentrations were conservatively treated as if they were entirely site-related in the risk estimates. Furthermore, given the temporal juxtaposition of site exposure and tissue sampling, there is little reason to believe that bone lead levels would be elevated relative to background when tissue lead levels are not elevated relative to background.

It should be clarified that bone and bone marrow are two different tissues. When discussing “bone” in this context, it is the mineralized (hard) portion of the bone. Bone marrow is part of the lymphopoietic system (lymphatics, blood, and blood forming tissue) and is related to bone only in its location in the body and in that it shares a name. While bone is a storage site for lead, bone marrow is not, and therefore it is important to discuss the two tissues separately.

Bone marrow is the more likely of the two tissues to be consumed. Bone marrow would not be expected to be preferentially enriched in lead relative to the organs sampled. In fact, because caribou bone marrow is more than 95 percent fat (Nutrition Data 2006), it is not a good source of minerals in general, and would be less likely to store the metals being evaluated at the site than the muscle and organ tissues that were sampled. In addition, bone marrow would make up an exceedingly small portion of the caribou tissue consumed by humans relative to muscle. Thus, because it is not a storage site and is a relatively small part of dietary intake, inclusion of bone marrow would have little or no impact on the results of the risk assessment. Nevertheless, collection of bone marrow will be considered during the development of the risk management plan.

Bone is a storage site for lead, and would be more likely to reflect very long-term exposure than soft tissues such as liver, muscle, and kidney. However, as with bone marrow, if bone consumption were included in the risk assessment, it would have little impact on overall risk results because bone would comprise a very small portion of the overall amount of caribou consumed by people, compared with muscle tissue. In addition, it is important to remember that the caribou metals concentrations used in the risk assessment come from caribou that over-wintered at the site. If site metals do affect metals concentrations in caribou, it would be reflected in the recent "exposure" experienced by these over-wintering caribou, and highly vascularized soft tissues such as liver should reflect that exposure.

The primary limitation in the Exponent (2002e) evaluation (see Appendix H) was the lack of access to data for individual animals for the 1996 study groups from Red Dog and elsewhere in Northern Alaska. Although the comparisons made using means and standard deviations consistently indicate a lack of difference between Red Dog and other areas, a statistical comparison using individual sample concentrations would further clarify this area of uncertainty.

5.4.3.10.2 Salmonberry and Sourdock Samples

Salmonberry and sourdock were sampled in summer 2004 from three traditional harvesting locations at increasing distance from the port facilities. Although samples were analyzed for the entire set of CoPCs for use in the risk assessment, this analysis focused on lead and cadmium, which were also the focus of a 2001 berry and sourdock investigation (ADPH 2001) and a subsequent analysis by Alaska Community Action on Toxics (ACAT 2004). Metals concentrations were compared between sites, to samples collected in 2001 from Ipiavik South (one of the 2004 sites), and to samples collected in 2001 from a reference location near Noatak.

These comparisons did not identify any significant differences in cadmium or lead concentrations in salmonberries and sourdock harvested from any of the three sites evaluated. In addition, metals concentrations were the same as or significantly less than reference concentrations. Results from the risk assessment indicate that the metals concentrations detected in salmonberries and sourdock collected at the site are not associated with elevated human health risks. However, the results of the comparison with metals concentrations in salmonberries from reference areas suggest that those risk estimates are more related to background exposure than to site-related metals.

The primary area of uncertainty in the salmonberry and sourdock subsistence food study is the potential variation in metals concentrations based on the temporal proximity of sampling and rainfall. It is possible that a rain event just prior to sampling could wash off dust that otherwise might have been included in the analyses, thereby potentially decreasing the detected metals concentrations. This uncertainty can be further evaluated in future sampling events as part of an ongoing monitoring program.

5.4.3.10.3 Ptarmigan

Ptarmigan were collected in 2004 from near the DMTS road and from reference locations and were analyzed for antimony, barium, cadmium, lead, thallium, and zinc in breast muscle, liver, and kidney tissues. Metals concentrations from near-road samples were compared with reference samples and with concentrations reported in the literature. Concentrations of antimony and thallium were lower than or equal to reference concentrations in all three tissues. In addition, antimony was never detected in muscle, kidney, or liver tissue, and thallium was never detected in muscle and infrequently detected in liver and kidney tissue. The comparison strongly suggests that there is no site-related increase for antimony or thallium.

Average concentrations of barium and zinc in near-road samples were somewhat elevated relative to reference averages for all tissue/metal combinations except zinc in muscle, but none of these findings were statistically significant. Lead concentrations in liver and kidney tissues from near-road samples were statistically significantly elevated relative to reference concentrations. Lead concentrations in DMTS-area samples were also elevated relative to limited data available in the scientific literature.

Although lead concentrations in liver and kidney appear to be elevated in ptarmigan tissues, the risk assessment indicates that overall metals concentrations are still quite low. Results from the risk assessment indicate that metals concentrations in ptarmigan collected from the site are not associated with elevated human health risks at the levels at which they are consumed by the community.

The primary limitation of the ptarmigan study is small sample size. In particular, only three animals were captured in the reference area. This limits the strength of the conclusions that can be drawn on the basis of the ptarmigan data alone.

Taken together, the results from the three subsistence foods investigations, in conjunction with the risk assessment, suggest that the risks associated with continued harvesting of subsistence foods from the site, including in unrestricted areas near the DMTS, are not significantly elevated.

6 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 occur at the site. The following sections quantify and interpret ecological risks for plant and invertebrate communities and wildlife populations that may be exposed to site-related CoPCs. In Section 6.1, the problem formulation, the results of the ecological screening assessment (presented 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. In Sections 6.2 through 6.4, risks to communities of lower-trophic-level organisms that may be exposed to CoPCs at the site, including invertebrates, plants, and fish, are assessed separately for terrestrial, freshwater aquatic, and coastal lagoon environments. Section 6.5 presents the risk assessment for wildlife (birds and mammals). In that section, deterministic 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. Uncertainties associated with the risk assessment are identified and discussed in Section 6.6, and in Section 6.7, risks to all receptors are evaluated by environment to determine the overall ecological significance of risk assessment results.

6.1 Problem Formulation

The problem formulation for the ERA draws upon the results of the screening assessment and the site-specific knowledge acquired through Phase I and supplemental (Phase II) sampling to refine the list of CoPCs and the preliminary conceptual model initially presented in Section 2.4. The problem formulation also describes complete exposure pathways, ecological receptors, and assessment and measurement endpoints to be evaluated. Complete exposure pathways and relevant receptors are integrated into a refined CSM in this section.

6.1.1 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 3-38 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 maximum 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.

6.1.2 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.

6.1.2.1 Terrestrial

Primary exposure pathways in the terrestrial environment include direct contact, uptake, and ingestion. Compared to the primary exposure pathways, inhalation of soil particles is considered a secondary exposure pathway that may represent minor exposures for birds and mammals, as discussed in Section 2.4.4. Therefore, in the terrestrial environment, the baseline assessment evaluates 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.

6.1.2.2 Streams

In streams, the baseline ERA evaluates 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. However, fish may be exposed to CoPCs in sediment or prey, and therefore risks to fish in freshwater systems are also considered in the baseline ERA.

The quantitative risk assessment for aquatic birds and mammals that forage in streams estimates 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 using conservative exposure and effects parameters, 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-level hazard quotients for avian invertivores exceeded 1.0 and were at least 2-fold greater than reference hazard quotients for some CoPCs; therefore, risks to these receptors are quantified in the baseline ERA. At the time of the CoPC screening, there were insufficient data to evaluate risks to herbivorous birds and mammals that may use streams and creeks, and therefore risks to these receptors are also evaluated in the baseline ERA.

6.1.2.3 Tundra Ponds

Risks to aquatic or wetland plants and herbivorous birds and mammals are assessed in the tundra pond environment, as there were insufficient data at the time of the CoPC screening to eliminate these pathways. Concentrations of several CoPCs in tundra pond sediment exceeded screening benchmarks for invertebrates and other aquatic life (Table 3-22), and screening-level food-web models for birds foraging in site ponds resulted in hazard quotients greater than 1.0

and greater than reference hazard quotients for some chemicals (Table 3-34). Exposure pathways to these receptors, however, may be incomplete in the tundra ponds based on results of the Phase I 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 I 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 is assessed in the baseline ERA. 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, risks to freshwater avian invertivores that feed on aquatic and terrestrial invertebrates around the fringes of tundra ponds are evaluated in the ERA, using terrestrial invertebrates as surrogates for tundra pond invertebrates. Avian invertivores may be exposed to CoPCs primarily through incidental ingestion of soil and ingestion of food.

Based on observations from the Phase I 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 I were hydrologically disconnected from surface water inputs from streams, and some were 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 I. Therefore, pathways to fish and piscivorous wildlife are considered incomplete in tundra ponds, and these receptors are not assessed in this environment in the baseline ERA.

6.1.2.4 Coastal Lagoons

Pathways to aquatic and wetland plants, aquatic invertebrates, and herbivorous and invertivorous wildlife exist in coastal lagoons, and risks to these receptors are assessed in the baseline ERA. 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, although fish may also be exposed to CoPCs in sediment or prey. Attempts were made to sample fish in Port Lagoon North, the North Lagoon, and the reference lagoons during the 2004 supplemental sampling program, but no fish were observed in the lagoons; baited minnow traps and beach seining failed to capture any individuals. Therefore, pathways to coastal lagoon fish are considered incomplete and are not assessed further. Pathways to piscivorous birds and mammals that may feed on fish in coastal lagoons are also incomplete and are not assessed quantitatively in the baseline ERA.

6.1.2.5 Coastal Marine

In marine sediment, concentrations of CoPCs were below the ERL screening criteria for all samples from both sampling events (pre-shipping and during-shipping) in 2004 (as described in Section 4.3). Based on these results, no CoPCs were identified for the marine environment.

6.1.3 Refined Conceptual Site Model

Based on the results of the ecological screening and the site-specific knowledge gained during Phase I 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 6-1, 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 6.1.2. Primary exposure routes are quantified in the ERA, while secondary exposure routes are addressed qualitatively in the uncertainty analysis.

6.1.4 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 I and Phase II sampling events. 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 five 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.
- Survival, growth, and reproduction of coastal lagoon mammalian:
  - Herbivore populations.

Because the screening results indicate that a baseline ERA is not warranted in the coastal marine environment, no assessment endpoints were retained for that environment. Table 6-1 summarizes the assessment endpoints for the ERA.

6.1.5 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 6-1.

The measurement endpoints used to evaluate the impacts to assessment endpoints such as the structure and function of plant and invertebrate communities are focused 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.

6.1.6 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 are evaluated in the risk assessment:

- Willow ptarmigan (terrestrial avian herbivore)
- Tundra vole (terrestrial mammalian herbivore)
- Caribou (terrestrial mammalian herbivore)
- Moose (terrestrial, stream, and coastal lagoon mammalian herbivore)
- Lapland longspur (terrestrial avian invertivore)
- Tundra shrew (terrestrial mammalian invertivore)
- Snowy owl (terrestrial avian carnivore)
- Arctic fox (terrestrial mammalian carnivore)
- Green-winged teal (stream and pond avian herbivore)
- Muskrat (stream, pond, and coastal lagoon mammalian herbivore)

- Common snipe (terrestrial and stream invertivore)
- Brant (coastal lagoon avian herbivore)
- Black-bellied plover (coastal lagoon avian invertivore).

6.1.6.1 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 is assessed. Risk of adverse ecological effects to birds and mammals that may feed on plants at the site is 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 the CAKR and Noatak National Preserve and is known to nest in these areas (Schroeder 1998). In 1981–1982 baseline studies, willow ptarmigan were 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 1999).

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 (WACH) 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). Between-year differences in occurrence can be pronounced because caribou migration routes can vary annually. DFG tracked seasonal ranges of the WACH based on telemetry data for satellite-collared individuals (DFG 2003c). Between July 2000 and June 2001 no caribou were recorded in the Wulik, Noatak, and Kivalina drainages, but recordings were common in these drainages between July 2001 and June 2002. A small percentage of the migrants may remain near the site throughout the winter. Census data from 1983−1999 for the region incorporating the Wulik, Noatak, and Kivalina drainages indicate that winter densities (November 1−March 31) range from 0 to 2.6 caribou/square mile (DFG 2003c). Although caribou may have a clumped distribution during the winter, these low densities indicate that it is very unlikely that more than at most a few hundred individuals of the WACH, totaling more than 430,000 individuals (DFG 2003c) would be present near the DMTS during the winter. Fewer caribou are observed at the site during the spring and summer than during the fall migration because the site is outside of the summer range and calving grounds (DFG 2003c). 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.). The moose is evaluated as a terrestrial, freshwater aquatic (stream), and coastal lagoon receptor in the risk assessment.

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 are 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 2-7).

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 the CAKR and Noatak National Preserve and is known to nest in both parks (Schroeder 1998). 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 1999).

The tundra shrew (also known as the arctic shrew) is found across northern North America (University of Michigan 1997) 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 1997; YDRR 2002). The tundra shrew eats a diverse diet of invertebrates such as beetles, worms, spiders, slugs, snails, and insect larvae (University of Michigan 1997). It is the default indicator species chosen by DEC to represent terrestrial mammalian invertivores for risk assessments conducted in the northwest ecoregion (DEC 1999).

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). Southern migrations or irruptions are more common in years of low prey abundance. 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.).

6.1.6.2 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 be exposed to chemicals from these sediments is assessed (Table 6-1). Risk of adverse ecological effects to birds and mammals that may consume freshwater plants at the site is assessed using the green-winged teal (*Anas crecca*), the muskrat (*Ondatra zibethicus*), and the moose (described in Section 6.1.6.1) as receptors representing freshwater herbivores. These receptors may be exposed to CoPCs in surface water, sediment, and their diet (Table 2-7).

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 the CAKR and Noatak National Preserve (Schroeder 1998). 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 1999).

The muskrat, a large, herbivorous rodent, occurs across mainland Alaska south of the Brooks Range (DFG 2003a). Muskrats are present in the CAKR 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 1999).

Risk of adverse ecological effects to birds that may feed on freshwater invertebrates at the site is 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, soil or sediment, and its diet (Table 2-7). The snipe is evaluated in the risk assessment as a stream receptor and as a receptor in the terrestrial environment (in lieu of tundra ponds). 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 the CAKR and Noatak National Preserve (Schroeder 1998). 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 1999).

6.1.6.3 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 is assessed in lagoons (Table 6-1). Risks of adverse ecological effects to birds that may feed on aquatic plants or invertebrates near the port site are assessed using the brant (*Branta bernicla*) as the receptor representing coastal avian herbivores and the black-bellied plover (*Pluvialis squatarola*) as the receptor representing coastal avian invertivores. Risks to herbivorous mammals in the coastal lagoon environment are assessed using the muskrat and moose (described in Sections 6.1.6.1 and 6.1.6.2) as representative receptors.

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 (including coastal lagoon) and lacustrine waters, wet meadows and marshes, and sedge-grass tundra environments at the site (Dames & Moore 1983a) and is known to nest in the CAKR (Schroeder 1998). 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 2002). 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 1999).

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 2002). It nests in shallow scrapes on dry tundra, gravelly plains, or in coastal marshes (DFG 2003d; Paulson 1995; Hebert 2002) and is known to breed in the CAKR (Schroeder 1998). 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 1999).

6.2 Terrestrial Assessment

The terrestrial assessment includes evaluations of risk to terrestrial plant communities and tundra soil fauna. Ecological risks to terrestrial plant communities are assessed through the analysis and interpretation of plant community data collected during the 2004 supplemental sampling program. In addition, ecological risks are also assessed through comparison of terrestrial plant tissue concentrations (collected from site and reference stations during the 2004 supplemental sampling event) with phytotoxicity thresholds reported in the literature. Plant tissues were not shaken or washed prior to analysis, as discussed in Section 6.2.2, *Plant Tissue Comparisons with Phytotoxicity Thresholds*.

A three-tiered statistical approach was used to evaluate plant community data: 1) comparison of site communities with reference communities; 2) correlation between distance from sources, environmental variables, and plant community parameters; and 3) ordination of plant community data using principal component analysis (PCA) and nonmetric multidimensional scaling (NMDS). In Section 6.2.3, *Risk Characterization for Terrestrial Plants*, results of the plant community analysis are integrated with the comparison to phytotoxicity reference data to assess the potential for site-related CoPCs to cause adverse effects to vegetation communities along the DMTS road corridor.

Risks to tundra soil fauna are not quantitatively assessed in the ERA. However, terrestrial invertebrates were sampled in 2004 to provide prey data for food-web exposure models for terrestrial invertivorous wildlife (see Section 6.5). A qualitative discussion of terrestrial invertebrate communities at the site is provided in Section 6.2.4.

6.2.1 Plant Community Surveys

Terrestrial plant communities in the DMTS road corridor were surveyed systematically to characterize the vascular plant community composition and moss and lichen abundance at increasing distances from the road. Plant communities near the port, along the DMTS road, and near the mine were surveyed. Representative reference locations for each community type were also identified and surveyed for comparison with the site survey locations. In addition, plant communities at site and reference coastal lagoons were surveyed to identify any differences in their community structure. Lagoon plant community data were evaluated with the terrestrial plant data in the statistical analysis described below. The lagoon plant community data are reported and discussed as part of the terrestrial plant assessment presented in this section, and are also evaluated in more detail along with other lagoon community data in the risk characterization for coastal lagoon vegetation in Section 6.4.2.

Plant community structural parameters such as individual species' canopy cover, frequency of occurrence, and total species richness were recorded at each terrestrial and coastal lagoon sampling station. Overall vitality of the vegetation community was assessed qualitatively through field observations. Plant community functional parameters (e.g., biomass, productivity, energy flow) were not directly evaluated in this investigation. Results of the structural analysis were used in the risk assessment to infer possible changes in function. Field sampling methods are summarized in Section 6.2.1.1 and are described in more detail in Appendix E. Appendix I includes the plant community data and narrative descriptions of the vegetation observed at each station.

6.2.1.1 Survey Methods

Vegetation surveys were conducted in terrestrial and coastal lagoon environments to assess differences in plant communities between site and reference stations and with distance from fugitive dust sources (i.e., the DMTS road and port facilities). Vegetation parameters that characterize the structure of the plant community, and which may also reflect functional attributes of the community, were recorded at each survey station. Plant communities were evaluated in a series of sample plots (microplots) established at 13 site stations and 4 reference stations in the terrestrial environment and at 2 site stations and 2 reference stations in the coastal lagoon environment. Terrestrial stations were aligned along four transects perpendicular to the DMTS road, in order to evaluate communities at various distances from dust sources. Transects were distributed along the length of the DMTS road, so that plant communities near the port, in the central portion of the site, and near the mine were represented in the surveys. Measured parameters included the percent cover of different vascular plant species in microplots, which reflects their relative dominance in the vegetation community; the frequency of occurrence of vascular plant species in microplots, which reflects their commonness in the community; and the total number of vascular plant species identified in the microplots, which represents the vascular plant species richness of the community. Moss and lichen were also evaluated in the surveys but were not identified to the species level. Tundra soil characteristics, including CoPC concentrations, pH, and total solids (percent of wet sample mass that is solid material), were measured in samples collected at each survey station, in order to relate changes in vegetation to fugitive dust deposition.

Terrestrial survey locations at the site included 10-m, 100-m, 1,000-m, and 2,000-m stations along one transect at the port (TT5), and 10-m, 100-m, and 1,000-m stations on two transects located along the DMTS road (TT3 and TT8), and on one transect near the intersection of the road with the mine's ambient air/solid waste permit boundary (TT6; Figure 4-1). The 2000-m station on transect TT6 was assessed qualitatively, without formal plant community surveys. Transect TT5 begins near the DMTS road and runs past the CSB road loop and beyond toward the north. Thus, the DMTS road, loop road, and CSB facilities represent sources of fugitive dust to this transect, and the samples collected at the 100-, 1,000-, and 2,000-m stations along TT5 (measured from the DMTS road) were actually 85, 450, and 1,430 m from the closest dust source, respectively. In addition, vegetation communities located at road transect TT2, near the port's ambient air boundary, and at mine transect TT7, oriented northwest (predominantly downwind) of the mine's ambient air/solid waste permit boundary, were assessed qualitatively, without formal plant community characterization.

The survey stations near the port were characterized by a coastal plain wet-to-mesic tussock tundra community (referred to hereafter as "coastal plain" community) that had a tall shrub component at stations near the road (shrubs were considered "tall" if they exceeded the height of sedge tussocks; Photographs 8 and 9). Reference station TS-REF-12, located south of the port area and east of the Control Lagoon, was selected for comparison with coastal plain stations (Figure 4-1; Photograph 10). Road stations were located in a foothills mesic tussock tundra community (referred to hereafter as "tundra" community; Photographs 11 and 12). Terrestrial reference stations TS-REF-5 (Photograph 13) and TS-REF-7, located in the Evaingiknuk River drainage south (predominantly upwind) of the DMTS transportation corridor, were selected for comparison with tundra stations (Figure 4-1). A tall shrub component was present in tundra communities at station TT8-0010 near the road (Photograph 14) and TS-REF-5 at the reference area. Plant communities along transect TT2 (assessed qualitatively) generally had comparable compositions to communities at coastal plain and tundra stations (although the 1,000-m station was near a riparian corridor and therefore had a unique plant community compared to other 1,000-m stations, with less prominent *E. vaginatum* tussocks, abundant willows, and diverse forbs and graminoids). Stations located near the mine (transect TT6) were characterized by a hillslope mesic open shrubland community (referred to hereafter as "hillslope" community; Photographs 15 and 16). Reference station TS-REF-11, located in the Evaingiknuk River drainage, was selected for comparison with hillslope stations (Figure 4-1; Photograph 17). Stations on transect TT7 (assessed qualitatively) were located on ridge tops in a dry alpine tundra community that differed in composition from all other community types evaluated (Photograph 18).

At each vegetation survey station, plant community parameters were measured in ten 1-m square quadrats (microplots) spaced evenly along a 300-ft line that roughly paralleled the DMTS road. In each microplot, percent cover of live tissue was estimated for each vascular plant species that occurred in the plot, using the cover classes shown in Table 6-2. Cover was estimated in two dimensions only, and therefore plant cover that was under the canopy of taller species was not captured in the estimate. Thus, the cover percentage for a plant species in a microplot may be considered an expression of its dominance in the community, with plant height as an important contributing factor. The "trace" cover class was selected for species with negligible cover, such as small species that occurred only once in a microplot, larger plants that occurred outside the microplot, but which had overhanging leaves that contributed to the plot's cover, and species that were completely shaded by the canopy of other species. In addition, cover classes were assigned for total moss, total lichen, broadleaf litter, dry graminoid blades, and inorganic substrates (e.g., bare ground) in each microplot. These covers were estimated independently from the vascular canopy cover and therefore included areas shaded by vascular plants or by other components of the plot. Mosses and lichens were not assessed at lower taxonomic levels but rather were treated as broad categories. Average percent cover values of vascular plant species and other categories such as moss were calculated for each station from the ten microplot canopy cover estimates. For calculation purposes, each cover class was converted to the midpoint value before averaging across microplots (e.g., 15 percent for 5–25 percent cover range). Trace covers were not included numerically in the average cover calculations. At stations where shrubs exceeded the height of sedge tussocks, canopy covers for shrubs were also estimated using the line intercept method (Barbour et al. 1980) along the 300-ft survey line. In general, there was good agreement between the line intercept results and the average microplot cover estimates for shrubs.

The frequency of occurrence of each vascular plant species was calculated for each station as the percentage of the ten microplots in which the species occurred. Plant species with trace covers were included in the frequency measures. Frequencies were also calculated for moss, lichen, broadleaf litter, dry blades, and inorganic substrate categories. Vascular plant species richness (total number of species identified in the ten microplots) was recorded for each station. Species that were identified in the vegetation community (within a few meters of the survey line) but were not represented in any microplots were also recorded to develop an "area richness" estimate.

Tundra soil parameters that may influence vegetation communities were measured in collocated soil samples collected at each vegetation survey station (for clarity, references to "tundra soil" are simplified to "soil" throughout Section 6.2). These measurements included CoPC concentrations, pH, and total solids.

At tundra community transect TT8, in addition to surveys conducted at 10-m, 100-m, and 1,000-m stations, canopy covers and soil parameters were also measured at intermediate distances from the DMTS road (every 50 m out to 800 m, plus a survey at 900 m) in order to evaluate changes in the plant community on a finer scale than the three- or four-station sampling method allowed. Only one microplot was evaluated at each of the intermediate distances, rather than the ten plots assessed at the typical 10-m, 100-m, and 1,000-m stations. Consequently, average percent covers and frequencies could not be calculated at those stations. The percent cover and frequencies for the individual microplots were used instead.

Four plant community surveys were also conducted along the inland shorelines of Port Lagoon North, the North Lagoon, the Reference Lagoon, and the Control Lagoon to evaluate potential effects to coastal lagoon fringe emergent communities (lagoon) from exposure to fugitive dust. Station CL-REF-1, located at the Reference Lagoon, was selected for comparison with station PLNL at the Port Lagoon North, and station CL-REF-2, located at the Control Lagoon, was selected for comparison with station NLK at the North Lagoon (Figure 4-4). Lagoon vegetation surveys were conducted using the same methods as the terrestrial plant community surveys, with the primary difference being that lagoon quadrats were oriented along a transect running parallel to the shoreline rather than the DMTS road. Photographs 19–22 show the vegetation survey lines at the four coastal lagoon stations. In addition, vegetation at station NLF, located on the ocean side of the North Lagoon (Figure 4-4; Photograph 23), was assessed qualitatively through field observations.

6.2.1.2 Statistical Methods

Statistical analysis of vegetation community data was conducted to investigate how the measured communities compare to their respective reference communities, to evaluate the relationship of community properties with distance from the road, and to evaluate how the differences can be characterized. Individual species data are highly variable; thus, average cover for vegetative types, or functional groups, was primarily used in the analyses. The functional groups used for average cover were forbs, graminoids, deciduous shrubs, evergreen shrubs, and unvegetated substrates (including bare ground, road gravel, and rock). Additionally, vascular species diversity, evenness, and richness were incorporated into the analyses.

The Shannon-Weiner index was used to calculate species diversity from vascular plant cover estimates (Barbour et al. 1980). This commonly used index is more sensitive to rare species (those with low frequency of occurrence) than many other diversity indices. The specific calculation is as follows:

$$H' = -\sum_{i=1}^{S} (p_i)(\log_2 p_i)$$

where:

S = total number of species that contributed to canopy cover

p_i = proportion of cover due to species *i*.

Diversity is one measure of the community but should be used in conjunction with other measures, including evenness and richness. Evenness is a measure of how even the species percent cover is. For example if one species comprises 90 percent of the cover and the remaining 20 species all have very low percent cover, totaling the remaining 10%, then evenness would be low. Another example with percent cover divided equally among the present species would have a high evenness value. For this study, Pielou's evenness index was used (Pielou 1966). This index has been systematically used and is sensitive to rare species (Beisel et al. 2003). Pielou's evenness index is calculated based on the Shannon-Weiner index as follows:

$$E_{pielou} = \frac{H'}{\log_2(S)}$$

where:

H' = Shannon-Weiner index

S = total number of species that contributed to canopy cover.

Species richness is a count of the total number of species present in the ten microplots evaluated per station.

Statistical analyses were conducted on these three indices (species diversity, evenness, and richness), average percent cover for the four vegetation functional groups (forbs, graminoids, deciduous shrubs, and evergreen shrubs), as well as average percent cover and frequency for moss and lichen.

Each community type, coastal plain, tundra, hillslope, and lagoon, was compared to its respective reference station using the Wilcoxon non-parametric test. The non-parametric test was used to avoid distributional assumptions about the data. Additional comparisons were made using combined communities to increase the sample size and thus increase the power of the test to detect differences between site stations and reference stations. Coastal plain and tundra communities were quite similar and thus were combined and tested against their corresponding combined reference samples. Also, these two communities showed similar changes with distance from the road, so samples were combined according to their respective distance. Samples were grouped into distance categories of greater than, less than, and at 100 m from the road, and each distance grouping was then compared to the combined reference samples. Each of the vegetation measures was compared, as well as each of the CoPC metals concentrations, pH, and total solids. Metals comparisons were tested using one-sided tests to determine whether concentrations were lower at reference stations. All other comparisons were made using two-sided tests. Table 6-3 provides the p-values for each comparison. A significance level of $p<0.10$, as opposed to $p<0.05$, was used to increase the likelihood of detecting differences (i.e., to increase the power of the test). This significance level was used throughout for all of the statistical analyses.

Next, each community measure was correlated with distance from the road, and results are presented in Table 6-4. Spearman rank non-parametric correlation was used because other correlation methods make assumptions regarding the distribution of the data that could not be tested reliably, given the small sample sizes. Non-parametric methods make no assumptions about the distribution of the data. Regression models were also fit to predict each community measure based on distance and $\log_{10}$-distance. Both distance and $\log_{10}$-distance were fit in order to evaluate impacts that may occur linearly away from the road as well as impacts that may drop off more quickly with distance from the road, or logarithmically away from the road. This analysis was done for all of the vegetation community measures as well as soil metals, pH, and total solids. The regression models used a $\log_{10}$-transform for soil metals concentrations, pH, and total solids. This was done to meet the method assumptions of normality and homogeneity of variance (equal variability across the range of input values). Additionally, Spearman rank correlations were calculated between all pairs of variables to better understand the community structure. The significant correlation estimates ($p<0.10$) are presented in Tables 6-5 and 6-6. These analyses were also conducted using only data from the coastal plain and tundra communities, in order to distinguish trends specific to the tussock tundra environment. Results are included in Tables 6-4, 6-7, and 6-8.

Ordination methods were used to better understand the community structure as a whole. PCA and NMDS are unbiased methods for describing the structure between many variables by creating new variables based on the interrelationships between the original variables. The PCA was run using only the vegetation variables (i.e., diversity, evenness, richness, forbs, graminoids, deciduous shrubs, evergreen shrubs, lichens, moss, vegetative litter, and a non-vegetated category). NMDS was run using the species level percent cover data, including subcategories for vegetative litter and non-vegetated cover. Using the new factor or axes variables from each analysis, correlations with distance, soil metals concentrations, pH, and total solids were estimated using Spearman rank correlation. The significant correlation estimates ($p<0.10$) are reported in Table 6-9 and 6-10.

Results of the statistical analyses are discussed in detail in the following sections.

6.2.1.3 Plant Community Survey Results

Percent cover and frequency results for each plant community are presented in Tables 6-11 through 6-14. These results include cover estimates and frequencies for vascular plant species and broader categories such as moss, lichen, plant litter, and unvegetated substrates (i.e., bare ground, road gravel, and rock). Moss and lichen cover assessments were conducted independently of the vascular plant canopy cover assessment, and thus average percent cover estimates for moss and lichen relate to the abundance of these groups rather than to their relative dominance in the community. Three indices of plant community structure and function, including species diversity, evenness, and richness, were calculated from the vascular species percent cover and frequency results (Section 6.2.1.2) and are summarized in Table 6-15. Table 6-15 also includes an estimate of vascular species richness in the general vicinity of the microplots (area richness), including plant species observed in the area but not captured in the microplots.

Vascular plant species identified in microplots were classified as forbs, graminoids, deciduous shrubs, or evergreen shrubs, and average species covers within each group were summed in order to evaluate broad-level changes in functional groups near the DMTS road. Figure 6-2 shows the compositions of the live vascular plant canopies at vegetation survey stations in the coastal plain, tundra, and hillslope communities. As the tallest plants in the survey area, deciduous shrubs such as diamondleaf willow (*Salix planifolia pulchra*), dwarf birch (*Betula nana*), and alpine blueberry (*Vaccinium uliginosum alpinum*) tended to dominate terrestrial plant communities based on percent cover estimates (Figure 6-2; Tables 6-11 through 6-13).

Table 6-16 presents the CoPC concentrations, pH, and total solids measured in soil samples collected from the terrestrial and coastal lagoon plant community survey stations. The CoPCs are those chemicals that could not be eliminated from the risk assessment based on comparisons with soil screening benchmarks and reference concentrations (see Section 3.6).

The following sections provide an overview of the coastal plain, tundra, hillslope, and lagoon plant communities, summarize field observations, and report the results of the statistical evaluations. Statistical comparisons between site and reference stations are presented for each community type and for the combined coastal plain and tundra stations, which were also grouped by distance from the DMTS road and port facilities, as described in Section 6.2.1.2. Subsequent sections describe plant community trends observed with distance from the DMTS road or port facilities, and highlight significant correlations between plant community variables and distance from dust sources. Significant correlations between all vegetation and soil variables are also reported. The final section presents the results of the vegetation PCA, along with correlation of the major PCA factors with distance from the DMTS road and soil variables.

6.2.1.3.1 Overview of Plant Communities

Coastal plain and tundra habitats had similar plant communities dominated by tussock-forming cottongrass (*Eriophorum vaginatum*) and dwarf shrubs, including dwarf birch, diamondleaf willow, alpine blueberry, salmonberry (*Rubus chamaemorus*), Labrador tea (*Ledum palustre decumbens*), lingonberry (*Vaccinium vitis-idaea*), and crowberry (*Empetrum nigrum hermaphroditum*) (Tables 6-11 and 6-12). A variety of mosses and lichens formed the bottom layer of these plant communities. Low-lying wet areas dominated by graminoids such as cottongrass (*Eriophorum angustifolium*) and water sedge (*Carex aquatilis*), and taller shrub complexes dominated by birch and willow, were also present in the tundra. The coastal plain community had the lowest total live vascular cover of the terrestrial communities surveyed. Reference stations in tundra (and hillslope) communities showed the highest total covers (Figure 6-2). The soil (organic horizon) pH measured at 1,000-m and 2,000-m stations, away from the maximum influence of the DMTS road, and at reference stations was less than 5.0, signifying an acidic environment (Table 6-16). The species compositions of the coastal plain and tundra communities were similar to the moist acidic tundra complexes common to the arctic foothills of northern Alaska (Walker et al. 2001; Walker 2000; Walker et al. 1994). The vegetation type maps presented in the baseline studies identified a tussock tundra community in the vicinity of the coastal plain and tundra transects (Dames & Moore 1983a).

The hillslope plant community was characterized by deciduous shrubs such as birch, blueberry, and several willow species (*Salix glauca*, *S. pulchra*, *S. reticulata*, and *S. lanata*), nontussock sedges such as *Carex bigelowii*, diverse forbs, and mosses and lichens (Table 6-13). The hillslope community lacked the tussock physiognomy of the coastal plain and tundra vegetation (*E. vaginatum* was only present at one hillslope station, TT6-0100), and evergreen shrubs and graminoids were generally less dominant in the hillslope community than in the tussock tundra habitats (Figure 6-2). Hillslope site stations were located at higher elevations (approximately 800–930 ft) than coastal plain (approximately 60–75 ft) and tundra (approximately 480–625 ft) site stations and had tundra soil pH values greater than 5.0 (Table 6-16). The hillslope plant community shared some characteristics with the tussock tundra communities, such as the dominance of birch in the shrub canopy, but aspects of its species composition (such as the presence of *Dryas integrifolia* and some basiphilous forbs) and the community's higher species richness more closely resembled moist nonacidic plant associations found in northern Alaska (Walker et al. 2001; Walker 2000; Walker et al. 1994). The vegetation type maps from the baseline studies identify the area as a mix of low shrub and sedge-grass tundra (Dames & Moore 1983a).

The dry alpine tundra communities on transect TT7 are adapted to wind exposure and rocky substrates. Dominant plants included dryas (*D. octopetala*), dwarf willows (*Salix phlebophylla* and *S. reticulata*), *Carex* sedges (*C. microchaeta*, *C. scirpoidea*, and *C. podocarpa*), and lichens. Shrubs such as dwarf birch, alpine blueberry, spirea (*Spirea beauverdiana*), Labrador tea, lingonberry, and heather (*Cassiope tetragona*) were dominant at station TT7-0010, which was situated in a shallow bowl on the lee side of a ridgeline, where snow likely lingers until late spring. Vegetation maps from the baseline studies classify the area as dwarf shrub mat and cushion tundra (Dames & Moore 1983a).

Coastal lagoon plant communities at stations PLNL, NLK, CL-REF-1, and CL-REF-2 consisted of wetland vegetation dominated by mare's tail (*Hippurus vulgarus*) and graminoids, including tundra grass (*Dupontia fischeri*), pendent grass (*Arctofila fulva*), *Carex* spp., and cottongrass (*E. angustifolium*; Table 6-14). Bryophytes formed the ground cover at most lagoon stations (Table 6-14). Hydrophytic vegetation along lagoon margins transitioned into mesic tussock tundra farther inland, as illustrated in Figure 6-3. The vegetation at station NLF was a coastal dune community dominated by beach wild rye (*Elymus arenarius mollis*) The baseline studies identify the vegetation around coastal lagoons as marsh vegetation with tall grass sand dune communities along the coast (Dames & Moore 1983a).

6.2.1.3.2 Summary of Field Observations

The overall vitality of vascular vegetation was assessed qualitatively at each survey station in the field. Along all terrestrial survey transects, heavy amounts of road dust were observed on plant foliage at 10-m stations (Photograph 24), and lighter amounts of dust were detectable by sight or touch at 100-m stations. Coastal plain transect TT5 and tundra transect TT8 were characterized as particularly dusty at 10-m and 100-m stations. On transect TT2, there was dust deposition on roadside vegetation and impounded water near the road (Photograph 25). Dust was not detected on plant foliage at 1,000-m stations or at terrestrial reference stations. Along transect TT8, which was sampled at 50-m intervals, dust was not evident on leaves beyond the 150-m station. Field notes indicate that gravel spray from snow plowing was observed in the tundra up to approximately 50 m from the road. On transect TT7 near the mine, vegetation was not visibly dusty. Coastal lagoon vegetation was not noticeably dusty at any site or reference station.

In some cases, shrubs appeared less healthy near the road; according to field notes, there were higher incidences of low leaf cover or bare branches, persistent dead leaves, brittle branches, and discolored foliage at 10-m and 100-m stations in comparison to more distant stations along terrestrial transects (Photograph 26). Sedge tussocks appeared to be taller and more robust at 1,000-m stations than at 10-m and 100-m stations in the tundra community. At station TT7-0010, located just outside the mine's ambient air/solid waste permit boundary, some shrubs appeared to be in poor condition (blackened, bleached, dry, or dead; Photographs 27 and 28). The station's position on the lee side of a ridge suggests that it is a deposition area for snow and possibly mine dust. Station TT7-0010 had the highest lead concentrations in tundra soil and lichen of any station sampled in the 2004 supplemental sampling program.

Occasionally, defoliation of shrubs such as dwarf birch and blueberry was also observed at terrestrial reference stations, although this generally appeared to be the result of herbivory. Brown or bleached foliage on evergreen shrubs such as Labrador tea, crowberry, and lingonberry was also seen at terrestrial reference stations (Photograph 29). In the vicinity of station TS-REF-5, bleached lingonberry was noted in a snow accumulation area. The field notes indicate that phenomena such as loss of deciduous foliage or brown or bleached evergreen shrubs seemed to be present at some level at most or all stations, regardless of whether dust was detected on vegetation. Discolored evergreen foliage may indicate drought stress or some other natural stressor, but it is very unlikely that effects are the result of metals deposition from the DMTS corridor, as metals concentrations in tundra soil were generally low at reference stations relative to site stations (Table 6-16). The height of sedge tussocks was variable at the reference stations; tussocks at station TS-REF-5 were shorter and less densely spaced than tussocks at site stations such as TT3-1000 or TT8-1000.

No vegetation anomalies were recorded for site or reference coastal lagoon communities. There were abundant signs of wildlife use at the Control Lagoon (station CL-REF-2), including bear scat, goose scat, clipped sedge blades (signs of grazing), and possible animal bedding areas.

The substrate characteristics and plant community composition at station NLF were similar to those of sand dune habitats at the reference lagoons.

Vitality of nonvascular flora was also assessed qualitatively through field observations. In general, mosses at 10-m and 100-m stations in the coastal plain and tundra communities appeared to be less robust and less diverse than in communities farther from the road or in reference areas. For example, the moss cushion at tundra stations TT3-0010 and TT3-0100 was described in field notes as thinner, drier, and less vivid than mosses at station TT3-1000. This effect was not very apparent in the hillslope community. Dust-laden moss that appeared to be dead was observed at TT3-0010 and TT8-0100 (Photograph 30). Mosses at site and reference stations in the coastal plain community appeared to be dry or bleached in some microplots; perhaps this effect is an exhibition of drought stress. The available information does not help to determine if bleached vegetation was more common near the site than at reference locations. Coastal plain stations were surveyed following periods of sunny and relatively warm weather, which may have contributed to the dryness in moss (and vascular plant foliage) noted in both site and reference plant communities at that time. Analysis of quantitative vegetation community parameters such as percent cover of litter (i.e., dry blades or broad leaf litter) is discussed in Section 6.2.1.3.6.

Mosses were not identified to lower taxonomic levels in the field program. However, qualitative observations along tundra transect TT8 (surveyed with a single microplot every 50 m) suggested that sphagnum species were more common and robust at greater distances from the road. Sphagnum was obvious in the community around the microplot by about 500 m from the road, and was first encountered within a microplot at the 700-m station. By 800 m from the road, sphagnum appeared robust. In contrast, close to the road, sphagnum species seemed to be absent or were not the dominant moss species in the community. Thus, in some locations along the DMTS road corridor, shifts in moss community structure may be occurring with distance from the road, but these changes were not documented systematically in the plant community surveys.

Lichens were difficult to find at 10-m and 100-m stations in coastal and tundra communities, and those that were present often looked discolored or crisp and dull-colored (Photograph 31). *Thamnolia subuliformis* was tentatively identified as the first lichen to enter the community as one moved away from the road, but its white thallus also made it one of the easiest to spot. Blackening of foliose lichens at hillslope station TT6-0010 was also observed. Lichens were obvious and more diverse farther from the road (Photograph 32), and were healthy-looking, abundant, and diverse at tundra reference stations. Similar to mosses, lichens at all coastal plain site and reference stations seemed dry. Lichens were abundant at station TT7-0010 (near the mine's ambient air boundary) but appeared dry and in some cases darkened or dead.

Representative photographs of vegetation at various distances from the DMTS road are provided for each of the three terrestrial plant communities assessed in the vegetation surveys. Photographs 33 through 36 show examples of microplots evaluated at coastal plain stations TT5-0010, TT5-0100, TT5-1000, and TT5-2000, respectively. For comparison, two microplots evaluated at coastal plain reference station TS-REF-12 are presented in Photographs 37 and 38. Photographs 39 through 42 show the vegetation assessed in microplots along tundra transect TT8 at distances of 10 m, 200 m, 600 m, and 1,000 m from the road, respectively.

Photographs 43 and 44 show typical microplots at tundra reference stations TS-REF-5 and TS-REF-7. Photographs 45 through 47 show representative microplots at hillslope stations TT6-0010, TT6-0100, and TT6-1000, respectively, and Photograph 48 presents a typical microplot evaluated at hillslope reference station TS-REF-11.

6.2.1.3.3 Site and Reference Comparisons

Statistical comparisons of site and reference plant communities are presented in Table 6-3. Within plant communities, data for all site stations, regardless of distance from the road, were pooled and compared to their respective reference results. In addition, to increase statistical power, data for coastal plain and tundra stations were grouped by distance from the road and compared against all coastal plain and tundra reference data.

Forb and graminoid covers were not significantly different between site and reference stations in any plant community (Table 6-3). Deciduous shrub cover was not significantly different between site and reference stations in the coastal plain community (Table 6-3), although cover was lowest at the reference station (20 percent as compared to 24–33 percent at site stations; Figure 6-2). This trend was attributable to consistently higher birch cover at all site stations and higher diamondleaf willow cover near the road (Table 6-11). Deciduous shrub cover in the tundra community was significantly lower at the site than in the reference area, reflecting lower salmonberry cover, especially away from the road, and lower alpine blueberry cover at transect TT8 (Tables 6-3 and 6-12). In general, the coastal plain and tundra communities had higher birch and willow cover results at the site than at reference stations, and salmonberry and blueberry cover results tended to be lower at the site than at reference stations (Tables 6-11 and 6-12). Deciduous shrub cover was not significantly different between site and reference stations in the hillslope community (Table 6-3). One deciduous shrub species (*Salix ovalifolia*) was present at only one lagoon station (CL-REF-1), where it did not contribute to canopy cover (Table 6-14).

When coastal plain and tundra data were combined, evergreen shrub cover was significantly lower at stations less than 100 m from the road than it was at the comparable reference stations (Table 6-3). Cover estimates for this group of stations ranged from zero (TT5-0100) to 11.3 percent (TT8-0010), compared to 28.8–37.0 percent cover at reference stations (Figure 6-2; Tables 6-11 and 6-12). Evergreen shrub cover increased with distance from the road, and site cover approached or exceeded reference cover by 100 m on tundra transect TT8, 1,000 m on tundra transect TT3, and 2,000 m on coastal plain transect TT5 (Figure 6-2). Evergreen shrub cover was not significantly different between site and reference stations in the hillslope community (Table 6-3), and evergreen shrubs were not present at lagoon stations (Table 6-14).

Moss cover comparisons between site and reference stations showed different trends in the coastal plain and tundra communities than in the hillslope community. Moss cover at most site stations was lower than at reference stations in the coastal plain and tundra communities (Figure 6-4). Site and reference comparisons were statistically significant for the tundra community and for the combined coastal plain and tundra communities (Table 6-3). In contrast, moss cover in the hillslope community was up to one-third higher at site stations than at the respective reference station (Figure 6-4), although differences between site and reference stations were not statistically significant (Table 6-3).

Moss cover in the coastal plain and tundra communities tended to increase to levels comparable to reference levels with increasing distance from the road (Figure 6-4). Moss cover approached or surpassed reference levels by 2,000 m from the road at coastal plain transect TT5 and by 1,000 m from the road at tundra transect TT8; at tundra station TT3-1000, however, moss cover was still slightly below the reference cover range (37.4 as compared to 45.5–52.3 percent; Figure 6-4).

Moss cover was not significantly different between the two site stations and the two reference stations in the coastal lagoon environment (Table 6-3). Although lagoon station PLNL had much lower average moss cover than its most comparable reference station (3.25 percent as compared to 50.3 percent at CL-REF-1), it also had much higher standing water (68.3 percent versus 0.25 percent) and mare's tail cover (65.3 percent versus 6.8 percent), illustrating that the two survey stations had a different moisture regime. Station NLK had moss cover similar to that of its reference station (CL-REF-2; Table 6-14).

The frequency of moss occurrence in microplots was not significantly different between site and reference stations in any plant community (Table 6-3). Mosses were ubiquitous in the tundra and occurred in almost all microplots examined during the field study, including microplots at coastal lagoon stations (Tables 6-11 through 6-14). The only stations where moss was not present in all ten microplots were coastal plain station TT5-0010 (nine plots), tundra station TT3-0100 (nine plots), and lagoon station PLNL (four plots).

Average lichen cover was significantly lower at site stations than reference stations in the tundra community and for all combined groups of coastal plain and tundra stations (Table 6-3). Lichen cover was not significantly different between site and reference stations in the hillslope community (Table 6-3). Lichen cover estimates did not reach reference levels along any vegetation survey transects (Figure 6-4). For example, lichen cover at coastal plain reference station, TS-REF-12, was 2-fold higher than the cover at TT5-2000, and covers at tundra reference stations were 2- to 4.5-fold higher than covers at TT3-1000 and TT8-1000. In the hillslope community, lichen cover at reference station TS-REF-11 was slightly (about one-fifth) higher than the cover at station TT6-1000 (Tables 6-11, 6-12, and 6-13). The frequency of lichen occurrence in microplots was significantly lower at the site than at reference stations for the combined coastal plain and tundra stations and for subsets of these stations located at 100 m and less than 100 m from the road (Figure 6-4; Table 6-3). Lichens were not found in microplots at 10-m stations in the coastal plain and tundra communities (Figure 6-4). Lichen frequencies increased with distance from the road in coastal plain and tundra communities, as described below; lichen frequency was 40–60 percent at 100-m stations and 90–100 percent at 1,000-m and 2,000-m stations in those communities (Figure 6-4). Lichen frequency was 90−100 percent at hillslope site stations compared to 80 percent at the hillslope reference station (Figure 6-4), though the difference was not statistically significant (Table 6-3). In all terrestrial plant communities, lichen frequencies at 1,000-m were equal to or greater than reference frequencies (Figure 6-4).

Plant litter was not significantly different between site and reference stations in coastal plain, tundra, or hillslope plant communities. Relatively high water cover at station TT3-0100 (Table 6-12) because of wet conditions at this station resulted in significantly higher unvegetated cover at 100-m stations than at reference stations in the tundra environment (Table 6-3).

Vascular species diversity was not significantly different between site and reference stations in any of the plant community comparisons (Table 6-3). Evenness at combined coastal plain and tundra site stations close to the road was significantly lower than at reference stations, but conversely, site evenness was significantly higher than reference evenness at distances of 100 m and greater than 100 m (Table 6-3). Species richness did not differ significantly between site and reference communities, except for combined coastal plain and tundra stations greater than 100 m from the road (Table 6-3), which had fewer vascular plant species (10–11 species) than reference stations (12–14 species; Table 6-15).

In all three terrestrial plant communities, CoPC concentrations at stations near the DMTS road were higher than at reference stations (Table 6-16). When all stations within a community type were compared against the comparable reference station(s), CoPC concentrations in soil were higher at the site than at reference stations in all community types (Table 6-3). Comparisons between distance groups and reference stations (combined coastal plain and tundra) showed that, in general, differences in soil CoPC concentrations were significant for stations near the road but not for stations greater than 100 m from the road (Table 6-3). In coastal plain and tundra communities, soil pH was significantly higher at stations up to 100 m from the road than at reference stations, but pH was not significantly different between site and reference stations in the hillslope and lagoon communities (Table 6-3). Total solids were elevated in soil at 10-m stations relative to reference stations in all terrestrial plant communities (Table 6-16), and total solids were comparable in site and reference lagoon soils (Table 6-16).

6.2.1.3.4 Relationships with Distance from the DMTS Road

Relationships between vegetation and soil parameters and distance from the DMTS road and port facilities are summarized in Table 6-4. Forb cover had a significant negative correlation with distance from the DMTS road (Table 6-4). The relationship was stronger when tested without the hillslope community data (correlation estimate of –0.710 as compared to –0.494; Table 6-4). The coastal plain communities close to the road had larger forb components as opposed to more distant stations and the corresponding reference station, which had no forb cover (Figure 6-2). Forbs such as coltsfoot (*Petasites* sp.) and Jacob's ladder (*Polemonium acutiflorum*) were present in 90–100 percent of microplots and contributed to canopy cover at stations TT5-0010 and TT5-0100 (Table 6-11). No forbs were identified in any microplots at stations TT5-1000 and TT5-2000 or at the reference station (Table 6-11).

In the coastal plain community, more graminoid species were represented in microplots at stations near the road than at more distant stations (as reflected by the presence or absence of numerical values in Table 6-11). For example, there were eight graminoid species present in microplots at stations TT5-0010 and TT5-0100, whereas there were four graminoid species present in microplots at station TT5-1000. Total graminoid cover was lower near the road (Figure 6-2). However, graminoid cover was not significantly related to distance from the DMTS road (Table 6-4).

The relationship between total deciduous shrub cover and distance from the road was not statistically significant (Table 6-4), but some community trends with distance were apparent. In the coastal plain community, deciduous shrub cover shifted from diamondleaf willow near the road to dwarf birch and alpine blueberry at 1,000-m and 2,000-m stations (Table 6-11). Total deciduous shrub cover decreased with distance from the road in the tundra community; salmonberry cover decreased with distance along transect TT3, and birch cover decreased (although blueberry cover increased) with distance on transect TT8 (Table 6-12). In the hillslope community, deciduous shrubs became more diverse away from the road, as willow species gained prominence in the community (Table 6-13). For example, grayleaf willow (*S. glauca*) was dominant in the hillslope community at station TT6-0100, while dwarf birch and blueberry were dominant at station TT6-0010 (Table 6-13).

Overall, evergreen shrub cover increased significantly with distance from the road (Table 6-4). This relationship was driven by trends in the coastal plain and tundra communities, as illustrated in Figure 6-2. Along tundra transects TT3 and TT8, the increase in evergreen shrub cover with distance from the road corresponded to a decrease in deciduous shrub cover (Figure 6-2). No consistent relationship between total evergreen shrub cover and distance from the road was observed in the hillslope community (Figure 6-2), but there was a shift in evergreen species composition from lingonberry, crowberry, and Labrador tea at station TT6-0010 to dryas (*D. integrifolia*) at station TT6-1000 (Table 6-13). Hence the relationship between evergreen shrub cover and distance from the DMTS road was stronger when tested without the hillslope community data (correlation estimate of 0.741 as compared to 0.589; Table 6-4).

Average moss cover in the coastal plain and tundra communities increased with distance from the road, with the exception of station TT5-0100, which had anomalously high moss cover (62.0 percent) relative to stations TT5-1000 and TT5-2000 (Figure 6-4). The increasing trend in moss cover was statistically significant for combined coastal plain and tundra communities (Table 6-4). Moss cover showed the reverse trend in the hillslope community, where cover decreased with distance from the road (Figure 6-4). Lichen frequency and cover increased significantly with distance from the road across the site (Figure 6-4), and the relationships were stronger when tested without the hillslope community data (correlations of 0.911 and 0.994, respectively, as compared to estimates for all communities of 0.717 and 0.595, respectively; Table 6-4).

Total unvegetated substrate had a significant negative correlation with distance from the road (Table 6-4). Unvegetated substrates had their highest frequencies and cover estimates at stations near the road, and they did not occur in many microplots evaluated at stations greater than 10 m from the road (Tables 6-11, 6-12, and 6-13). Plant litter was not significantly correlated with distance from the road (Table 6-4).

Vascular species diversity did not correlate significantly with distance from the road (Table 6-4). Evenness increased significantly with distance from the road in the combined coastal plain and tundra communities (Table 6-4), but evenness did not exhibit a consistent trend with distance in the hillslope community (Tables 6-4 and 6-15). Vascular species richness decreased significantly with distance from the road in the combined coastal plain and tundra communities (Table 6-4). This trend was most pronounced at coastal plain transect TT5, where opportunistic forb and graminoid species had colonized disturbed areas near the road, and hydrophytic plants had established in wet areas near the road prism (Photograph 25). These species tended to drop out of the community at greater distances from road-related disturbance (Table 6-11). In the hillslope plant community, species richness varied from 25 species at station TT6-0010 to 23 species at station TT6-0100, to 38 species at station TT6-1000 (Table 6-15).

Chemical concentrations in soil were highest near the road and tended to decline substantially by 1000 m from the road (Table 6-16). Figure 4-13 displays the lead gradient in soil along tundra transect TT8, located in the central portion of the DMTS road. On this transect, lead concentrations in soil were elevated in the first 150 m from the road but decreased almost an order of magnitude by 300 m from the road (Figure 4-13[a]). Other metals show a similar pattern to lead (Figure 4-13[b]). Concentrations of most CoPCs decreased significantly with distance from the road (Table 6-4). Tundra soil pH also decreased significantly with distance from the road (Table 6-4), although this trend was not apparent in the hillslope community (Table 6-16). Along tundra transect TT8, soil pH first dropped below 6.0 at the 600-m station, dropped below 5.0 at the 750-m station, and reached the upper end of the reference range (3.9–4.5) at the 1000-m station (Figure 4-13). Total solids also decreased with distance from the road (Table 6-4).

Differences in slope, aspect, and elevation among plant community survey stations were most prominent in the hillslope community, where plant species composition and community indices (e.g., species diversity) appeared to be associated with the topographical pattern of the transect rather than trending strictly with distance from the road. Relationships between plant community variables and distance from the road tended to be stronger when tested without the hillslope community data (Table 6-4), indicating that environmental factors such as aspect or substrate characteristics may have had a more dominant influence over vegetation characteristics on the hillslope community transect than on the coastal plain or tundra community transects. The role of environmental factors in the hillslope community is discussed further in Section 6.2.3.3.

6.2.1.3.5 Correlations Between All Variables

Significant relationships between plant functional group covers, vegetation community indices, and tundra soil variables are summarized for all plant communities (including coastal plain, tundra, hillslope, and lagoon data) in Tables 6-5 and 6-6, and for combined coastal plain and tundra communities only in Tables 6-7 and 6-8. Many vegetation variables correlated significantly with CoPC concentrations, pH, and total solids. Forb cover had significant positive correlations with soil CoPC concentrations and pH, whereas evergreen shrub cover, lichen frequency, and vascular species evenness had significant negative correlations with soil variables (Tables 6-5 and 6-6). Although graminoid cover, deciduous shrub cover, moss cover and frequency, lichen cover, unvegetated substrate cover, and vascular species diversity and richness did not relate as strongly to soil variables, trends were consistent (Tables 6-5 and 6-6). As shown in Tables 6-5 and 6-7, concentrations of many CoPCs were positively correlated with pH and total solids. The strong intercorrelation of these soil variables reflects their relationships with distance from the DMTS road (Table 6-4).

When hillslope community data were excluded from the analyses, some correlations strengthened. For example, relationships with lichen cover and frequency tended to be stronger in the combined coastal plain and tundra communities (Table 6-8) than for all plant communities (Table 6-6). Moss cover had significant relationships with soil variables in the combined coastal plain and tundra communities (Table 6-8), whereas the relationships were not significant for all plant communities (Table 6-6).

6.2.1.3.6 Multivariate Ordination Analyses

The PCA results confirm the overall differences among the four vegetation groups (forbs, graminoids, deciduous shrubs, and evergreen shrubs), the differences within vegetation communities related to distance from the DMTS road, and the differences between site and reference survey stations described in previous sections. Figure 6-5 shows terrestrial and lagoon vegetation stations as they relate to the two most significant factors derived in the PCA after a Varimax rotation. Rotation of the factors eases interpretation of the results by more heavily weighting fewer variables per factor. This figure includes a table showing the standardized factor coefficients for each of the factors after rotation and their respective eigen values and explained variability both before and after rotation. Sixty-five percent of the variability in all eleven of the broad-level plant community variables can be explained by these two PCA factors (Figure 6-5).

The NMDS results confirm these same distinctions between the vegetation community types as well as their relation with distance from the DMTS road. The NMDS method was used with the individual species and non-vegetation percent cover values, as opposed to PCA, because this method is more robust to the spotty nature of the data. Because the lagoon, coastal plain, tundra, and hillslope communities evaluated in this study naturally had different species compositions, many species with measurable cover in one community were not present or were present in trace amounts (with cover values equal to zero) at some or all stations in another community, resulting in a patchy data set. NMDS analysis used a Bray-Curtis similarity matrix based on standardized variables. Figure 6-6 shows the first two axes of the NMDS results, and Table 6-10 presents the weight of each species or other cover category relative to the two axes. A Monte Carlo analysis of stress values for a range of dimensions supported interpretation of only the first two axes.

Both analyses separate stations by plant community, segregating lagoon stations, hillslope stations, and coastal plain and tundra stations into three distinct groups (Figures 6-5 and 6-6). In both analyses, coastal plain and tundra stations tended to cluster together, reflecting the similarities between the two communities.

In the PCA, Factor 1 separates lagoon stations based on their low species richness, high graminoid cover, and lack of deciduous shrubs and lichen (Table 6-14). Lagoons are also higher in non-vegetated cover, because more area was covered by water. Station PLNL is isolated from the rest of the lagoon stations because of its very high covers for mare’s tail (*Hippuris vulgaris*) and water. The hillslope community is distinct because of its high species richness, high deciduous shrub, lichen, and moss covers, and low graminoid cover (Table 6-13). This pattern is the reverse of the lagoon community.

NMDS Axis 1 separates the lagoon stations primarily based on a few graminoid species (*Calamagrostis deschampsiodes*, *Dupontia fischeri*, *Deschampsia caespitosa*, *Carex canescens*, *Arctophila fulva*) and forbs (*Ranunculus hyperborealis*, *R. confervoides*, *H. vulgaris*, *Potentilla egedii*, *Rumex arcticus*, *Stellaria crassifolia*) that were only present in this community (Table 6-14). Additionally, the sand and gravel category was found only at lagoon stations, along with significant detritus/fines and littoral matter. NMDS Axis 2 distinguishes the hillslope stations from the coastal plain and tundra stations (Figure 6-6). Table 6-10 shows that a combination of graminoid, forb, and willow species that occur predominantly or exclusively in the hillslope community is largely driving the separation of the hillslope stations from the other terrestrial stations. Station TT6-1000 is particularly isolated based on its rich forb community and the presence of other species that were unique to this station (e.g., *Carex saxatilis* and *Cassiope tetragona*).

The coastal plain and tundra plant communities are more similar to one another than to the other two communities. Characteristics of both these communities include relative evenness of species, generally high evergreen shrub cover, and high vegetative litter, as shown in the PCA results (Figure 6-5).

Additional distinctions within these three transects relate to distance from dust sources and thus show a gradient along each transect in both analysis results. In the PCA, Factor 2 separates stations based on distance from dust sources (Figure 6-5). Forb cover and unvegetated cover decrease with distance from dust sources, while evenness, litter, and evergreen shrub cover increase (Figure 6-5 and Tables 6-11 and 6-12). Although moss and lichen covers also increase with distance from dust sources (Tables 6-11 and 6-12), the rotated Factor 2 coefficients for these variables are low in absolute magnitude, and therefore differences in total moss and total lichen covers do not appear to be driving the separation of coastal plain and tundra stations according to distance. The greatest differences in plant communities were observed from 85 m to 450 m from dust sources on the coastal plain transect and in the first 100 m from the road along the tundra transects. Stations located farthest from dust sources converge in the PCA, although they are shifted along Factor 1 relative to the corresponding reference stations. Higher deciduous shrub and lichen covers at the reference stations may explain the shift.

The species level data from the NMDS results illustrate the same pattern (Figure 6-6). Axis 2 separates coastal plain and tundra stations according to distance from dust sources. Based on the NMDS analysis, coastal plain stations TT5-0010 and TT5-0100 are the most different from other coastal plain and tundra stations; these stations are high in unvegetated cover such as bare ground and road gravel, and are characterized by forb and graminoid species that did not provide measurable cover at other coastal plain or tundra stations at the site (e.g., *Anemone narcissiflora*, *Polemonium acutiflorum*, *Stellaria laeta*, *Valeriana capitata*, *Arctagrostis latifolia arundinaceae*, *Poa lanata*). Stations TT3-0010 and TT8-0010 are distinguished from other tundra stations in part by high *Rubus chamaemorus* and *Salix pulchra* covers, respectively (Table 6-12). Stations located farthest from dust sources converge with reference stations in the NMDS analysis. These stations have lower forb and unvegetated covers and higher moss, lichen, and evergreen shrub covers (e.g., *Ledum palustre* and *Vaccinium vitis-idaea*), and are missing the opportunistic forbs, graminoids, and willows that were found at stations near dust sources (Tables 6-11 and 6-12).

To understand better the relationships between the vegetation community characteristics, represented by the PCA factors and NMDS axes, and the environmental variables, these new variables were correlated with distance from the road and tundra soil characteristics, including CoPC concentrations, pH, and total solids. Correlations for only the coastal and tundra community data were of primary interest although overall correlations for all communities combined were also analyzed. The results are summarized in Table 6-9 for the PCA factors and Table 6-10 for the NMDS axes.

Factor 1 of the PCA, which characterizes differences among the plant communities (Figure 6-5), did not correlate significantly with distance from the road or other soil parameters. Factor 2, which generally characterizes the differences within plant communities with distance from the road (Figure 6-5), showed a significant positive correlation with distance from dust sources and significant negative correlations with pH, total solids, and most CoPCs (Table 6-9). Thus, stations that had positive values for Factor 2 (low forb cover and unvegetated cover, and high evenness, litter, and evergreen shrub cover) tended to occur further from the DMTS road and its influences (higher metals, pH, and total solids), while stations that had lower values for Factor 2 tended to be located closer to the road. Factor 1, which captures differences related to moss and lichen covers, did not have significant relationships with distance from the road, metals, soil pH, or total solids (Table 6-9). The same conclusions can be drawn from the correlations using data from all of the vegetation communities.

Correlations between NMDS axes and distance and soil parameters were strongest for Axis 2 for the coastal plain and tundra data. Correlations using data from all the plant communities combined were mostly not significant. Axis 2 was significantly negatively correlated with zinc concentrations in soil and significantly positively correlated with distance from dust sources and molybdenum concentrations (Table 6-10). Axis 2 was also negatively correlated with other metals, soil pH, and total solids, but the correlations were not significant. The most negative values for Axis 2 are associated with the coastal plain stations TT5-0010 and TT5-0100, with values increasing with distance from dust sources. Thus, as Axis 2 values increase because of increasing evergreen moss, lichen, and evergreen shrub covers and decreasing forb and unvegetated covers, distance from dust sources increases, and soil metals concentrations decrease.

6.2.2 Plant Tissue Comparisons with Phytotoxicity Thresholds

Unwashed terrestrial plant tissues collected at site and reference stations during the 2004 supplemental sampling program were compared against available phytotoxicity thresholds for vascular plants (McBride 1994; Langmuir et al. 2004; Davis et al. 1978). The thresholds represent the chemical concentrations in leaves and shoots that corresponded to observations of phytotoxicity, such as reduced growth or induced chlorosis. Test species tended to be agricultural crops or other plants adapted to temperate environments, rather than arctic species, and thus the thresholds derived from these studies are not specific to the types of plants collected along the DMTS road corridor. Tables 6-17 and 6-18 summarize the results of the comparisons for willow (*Salix* sp.) and dwarf birch leaves, and sedge blades (*E. vaginatum*), respectively. As plant tissues were not shaken or washed prior to analysis, the chemical concentrations reported in Tables 6-17 and 6-18 include metals on the external surface of the plant tissues, such as metals in dust that settled on the foliage, in addition to metals that were within the tissue itself and therefore available to the plant. As such, the concentrations are conservative estimates of actual metal levels in tissues.

Aluminum, cadmium, cobalt, and zinc concentrations in shrub leaves (willow or birch) exceeded their literature phytotoxicity thresholds at one or more site stations. Aluminum and cadmium concentrations in willow leaves from stations TT2-0010, TT3-0010 (aluminum only), TT5-0010, and TT8-0010 exceeded the lowest thresholds (Table 6-17). Aluminum and cadmium concentrations in shrub leaves at corresponding 100-m and 1,000-m stations did not exceed the phytotoxicity thresholds. Cobalt concentrations in willow leaves from reference station TS-REF-5 and site stations TT3-0100, TT8-0100, and TT8-1000 exceeded the lowest threshold value (Table 6-17). However, the highest cobalt concentration in shrub leaves was measured at reference station TS-REF-5 (8.03 mg/kg dry wt.; Table 6-17). Zinc concentrations in birch leaves from reference stations TS-REF-7 and TS-REF-11, and concentrations in all site samples (except for willow leaves from station TT6-1000) exceeded the lowest phytotoxicity threshold (Table 6-17). When compared against the highest minimum threshold for zinc (500 mg/kg dry weight), only zinc concentrations in willow leaves from stations TT2-0010 and TT5-0010 near the DMTS port exceeded the thresholds (Table 6-17). Shrub leaf concentrations of all other CoPCs were below the minimum phytotoxicity thresholds (Table 6-17).

For sedges, only aluminum and zinc concentrations in sedge blades exceeded the lowest phytotoxicity thresholds. Aluminum concentrations at stations TT2-0010, TT3-0010, TT5-0010, and TT8-0010 were up to 2-fold higher than the minimum threshold for aluminum (Table 6-18). Zinc concentrations at station TT5-0010 near the port and station TT7-0010 near the mine's solid waste boundary were also up to 2-fold higher than the minimum threshold (Table 6-18). Sedge blade concentrations of all other CoPCs were below the minimum phytotoxicity thresholds (Table 6-18).

In addition to vascular plant tissue samples, moss samples (*H. splendens*) were collected along DMTS port and road transects and analyzed for metals concentrations as part of the Phase I sampling program in 2003. Moss samples (*H. splendens*) were collected for metals analysis from stations in the vicinity of transect TT6 during a site characterization study in 2001. These data (unwashed samples) are provided in Appendix C. Also, in the supplemental sampling program in 2004, *Peltigera* and *Cladina* lichens were collected at terrestrial transects TT2 and TT5 near the port, TT3 and TT8 in the central portion of the DMTS road, and TT6 near the mine. Metals data for these unwashed lichen samples are provided in Appendix G. Copper and zinc concentrations in moss and lichen tissues were compared against sensitivity thresholds found in the literature. Folkeson and Andersson-Bringmark (1988) studied the effects of metals released from a brass foundry near Gusum, Sweden on the area's coniferous woodland vegetation, and from their findings, the authors proposed phytotoxicity thresholds for copper and zinc in a variety of moss and lichen species. The authors reported the following tissue threshold concentrations for dominant mosses and lichens, including epiphytic lichens (mg/kg dry weight in unwashed samples): 25–60 copper and 150–290 zinc in mosses, and 80–300 copper and 480–1,300 zinc in lichens, for first signs of reduction in cover; 35–90 copper and 190–350 zinc in mosses, and 100–600 copper and 550–1,800 zinc in lichens, for obvious reductions in cover; and 70–110 copper and 300–400 zinc in mosses, and 350–1,000 copper and

600–2,200 zinc in lichens, for apparent survival thresholds (Folkeson and Andersson-Bringmark 1988).

Based on these sensitivity ranges for moss, copper concentrations in moss near the port and along the DMTS road were below effects thresholds (copper concentrations were not measured in moss samples from transect TT6). Zinc concentrations in moss were potentially high enough to cause mortality in mosses up to 100 m from the road and up to 1,000 m from port facilities, and to cause reductions in cover up to 1,000 m from the road (Table 6-19). On transect TT6, zinc concentrations in moss up to 2,000 m from the road were high enough to be potentially toxic to the moss.

In the port and road areas, zinc concentrations in lichens were potentially high enough at 10-m stations and some 100-m stations to result in reductions in cover or even mortality, but concentrations were below toxicity thresholds for lichens at 1,000-m stations (Table 6-20). Transect TT5 was an exception; lichen concentrations were above toxicity thresholds at station TT5-1000 (450 m from sources) but were below them at station TT5-2000 (1,430 m from sources). At all stations along transect TT6, zinc concentrations in lichens were below toxicity thresholds for lichens reported by Folkeson and Andersson-Bringmark (1988). Copper data were not available for lichens along the DMTS road.

6.2.3 Risk Characterization for Terrestrial Plants

In this section, site and reference comparisons of vegetation communities, trends in plant community structure with distance from the DMTS road, and relationships between vegetation parameters and environmental variables are evaluated to determine the nature and extent of effects to vegetation in the DMTS road corridor. Comparisons of CoPC concentrations in plant tissues with phytotoxicity thresholds reported in the literature are also considered as part of a weight of evidence approach. Risks to coastal plain and foothills mesic tussock tundra communities, which occur near the port and along the majority of the road, and risks to hillslope mesic open shrubland located near the mine, are evaluated separately in the following subsections. A supplemental evaluation of port site vegetation is also presented. Risks to coastal lagoon fringe emergent communities are characterized in Section 6.4.2.4.

6.2.3.1 Coastal Plain and Foothills Mesic Tussock Tundra

Vegetation community survey results indicate that coastal plain and tundra plant communities within 100-m of the DMTS road are different from reference conditions and from stations farther away from the road, and qualitative assessments of plant vitality near the road tend to support this finding. Plants within 100 m of the road were visibly dusty (Photograph 24), or felt gritty, and some plants displayed signs of stress, such as tissue discoloration or defoliation (Photographs 26, 30, and 31). In general, plant communities in this distance interval had lower evenness, evergreen shrub cover, moss cover, and lichen cover than reference communities or communities at stations located farther from the road (Figures 6-2 and 6-4; Tables 6-12 through 6-15). Coastal plain transect TT5 and tundra transect TT8 also had higher forb cover and tall shrub cover at stations near the road (Figure 6-2; Tables 6-11 and 6-12). Evenness, evergreen shrub cover, moss cover, lichen cover, and lichen frequency all increased significantly with distance from the road in combined coastal plain and tundra communities, while richness, forb cover, and unvegetated substrate cover decreased significantly with distance (Table 6-4).

When data for coastal plain and tundra stations were grouped together, representing the variability in tussock tundra from the coastal plain up into the foothills nearer the mine, and were compared against combined coastal plain and tundra reference stations, vascular plant functional group covers were not significantly different at stations greater than 100 m from the road than at the reference stations (Table 6-3). Moss cover, though generally lower at site stations, was not significantly different from reference cover at this distance from the road, nor was lichen frequency (Table 6-3). Of the functional groups, only lichen cover was significantly lower at 1,000-m stations at the site (including station TT5-2000) than at reference stations (Table 6-3), suggesting that lichen abundance may not reach typical levels by this distance from the road. Vascular species richness was lower by two to four species at 1,000-m and 2,000-m stations than at reference stations (Tables 6-12 through 6-15), although the differences may be artifacts of sampling, as discussed in the uncertainty section (Section 6.6.2). Species that were present in microplots at reference stations but not in microplots at 1,000-m and 2,000-m stations tended to have trace covers or low cover values. Consequently, vascular species evenness was significantly higher at these site stations than at the reference stations (Table 6-15). Vascular species diversity was not significantly different between these site and reference stations (Table 6-15).

It is difficult to determine from this study which road-related factors account for the differences observed in the coastal and tundra plant communities, as the relationships are confounded. Environmental variables such as CoPC concentrations and pH were significantly correlated with distance from the road and with each other, as shown in Tables 6-7 and 6-8. The DMTS road exerts physical influences on the tundra within the first 100 m, altering the moisture regime and substrate composition near the road, and the structure of the vegetation community appears to have shifted to a complement of species better adapted to the altered conditions. The road impounds water at its base, creating moist microhabitats near the road prism. Forb and graminoid species adapted to moist conditions, such as *E. angustifolium* (cottongrass), were observed growing in stands near the road (Photograph 25). The increased moisture and disturbance near the road may contribute to the higher forb cover and higher forb and graminoid richness (number of species present) recorded at stations TT5-0010 and TT5-0100 (Table 6-11). For example, there were nine forb species present in microplots at station TT5-0010, whereas there were four forb species present in microplots at station TT5-0100. Road gravel and fines deposited by surface water runoff and passing vehicles increase the mineral content of the tundra soil, as evidenced by the high total solids measured in tundra soils near the road (Table 6-16) and the road gravel cover observed in 10-m microplots (Tables 6-11 and 6-12). Walker (1996) draws parallels between macrosite anthropogenic disturbances, such as roads, and their natural analogs, gravel bars in floodplains and talus slopes. The increased dominance of diamondleaf willow and tall polar grasses near the road may reflect a community shift toward species better adapted to habitats such as riverbanks or hillslopes. The presence of some vascular plants may also be related to the low moss cover near the road, as thick moss cover can prevent plant roots from reaching the soil, discouraging the germination and establishment of some vascular plants (Gough et al. 2000).

Road dust deposition is a regional phenomenon akin to windblown loess from river channels (Walker 1996). Calcareous road dust may raise the surface soil pH and enrich the tundra with nutrients such as calcium and magnesium (Walker 1996). Along the DMTS road corridor, dust was visible or detectable by touch on foliage at all 10 m and 100m stations and at stations up to 150 m from the road along tundra transect TT8 (Photograph 24). Alkaline dust from the road bed material (pH 8.4 at material site MS9) is likely contributing to the elevated tundra soil pH measured at 10-m and 100-m stations (Table 6-16). Figure 4-13 indicates that the tundra soil pH is elevated above reference values (3.6–4.5) well beyond 100 m in the tussock tundra, and that tundra soil pH may not stabilize until nearly 1,000 m from the road. In addition, zinc and lead concentrates have pH values ranging from 7.5 to 8.5 (Teck Cominco 2003b,f), and calcium chloride, applied to the road as a dust suppressant, has a pH ranging from 7 to 10 (Tetra 1998). Therefore fugitive dust may contain concentrates, road bed materials, and calcium chloride, all of which may be contributing to elevated soil pH in tundra surrounding the DMTS road and port facilities.

Calcareous road dust is known to reduce total plant biomass and to affect species composition in acidic tundra (Auerbach et al. 1997). Studies of tussock tundra along the Dalton Highway, a gravel road on the North Slope of Alaska, have shown trends in plant community composition with distance from the road similar to some of the trends observed in the DMTS road corridor. These include stressed ericaceous shrubs (including Labrador tea and blueberry) in highly dusty areas; significantly lower covers near the road for evergreen shrubs such as Labrador tea and lingonberry; significantly lower lichen covers near the road; and desiccated mosses and significantly lower moss covers, particularly for *Sphagnum* spp., near the road; (Walker and Everett 1987; Auerbach et al. 1997). However, other plant community trends observed along the Dalton Highway, including lower vascular species richness and higher *E. vaginatum* cover near the road (Auerbach et al. 1997), were not observed in the DMTS road corridor (Table 6-15; Figure 6-2).

Plants with mat or prostrate growth forms, evergreen plants, mosses, and lichens seem to be the plant types most vulnerable to road dust effects (Auerbach et al. 1997; Walker and Everett 1987). These were also the groups with the most notable cover differences near the DMTS road in the coastal plain and tundra communities (Figures 6-2 and 6-4; Tables 6-11 and 6-12). Dust deposition may have a larger cumulative impact on shrubs that trap dust or retain their leaves from year to year than on erect deciduous shrubs (Auerbach et al. 1997). Dust on leaves may adversely affect plants by interfering with gas exchange, blocking light for photosynthesis, and absorbing radiation, which raises leaf temperatures (Spatt and Miller 1981). Mosses and lichens absorb water and nutrients from the air and are therefore susceptible to atmospheric deposition (Auerbach et al. 1997). Lichens may be eliminated entirely in areas with high dust and appear to be the most affected growth form in the tundra, particularly soil lichens *Cladina* spp. and *Peltigera* spp. (Walker and Everett 1987). Mosses lack a protective waxy cuticle on their leaves and stems, and dust that settles on these surfaces may absorb water from them, drying out the moss (Spatt and Miller 1981). Sphagnum mosses, normally dominant in moist acidic tundra, are best adapted to acidic, low-calcium environments (Spatt and Miller 1981), and conditions with high pH and high calcium are lethal to some *Sphagnum* species (Clymo 1973). Sphagnum moss has a high cation exchange capacity and is able to acidify its surroundings (Clymo 1973; Vitt et al. 1988); it also retains moisture, insulates the permafrost, and creates microtopography in the tundra (Auerbach et al.1997). Therefore, sphagnum may be considered a keystone plant of moist acidic tundra, and its disturbance may have a cascading effect on tundra plant communities. Along the Dalton Highway, weedy, minerotrophic mosses replaced *Sphagnum* species near the road, although moss cover was lacking entirely in the dustiest areas (Auerbach et al. 1997; Walker and Everett 1987). Alkaline road dust and calcium chloride applied for dust control may also create conditions unfavorable to sphagnum moss along the DMTS road. Qualitative assessments of tundra vegetation suggested that sphagnum was not common close to the road. However, covers of individual moss species were not assessed as part of the plant community surveys, and further study would be required to identify shifts in moss species composition with distance from the DMTS road. Other changes that may occur as a result of the presence of the road include physical effects on hydrology, snow accumulation downwind of the road prism, early melting of snow near the road as a result of increased albedo from dust on the snow, and a deeper thaw of permafrost in these areas. These physical changes contribute to plant community changes along any road, independent of any potential effects from metals in the deposited dust.

Fugitive dust from the DMTS road and port facilities also contains elevated metals concentrations that may be affecting vegetation communities in the road corridor. Phytotoxicity was not evaluated directly in this study, other than the qualitative observations made during sampling. However, CoPC concentrations measured in plants can be compared with effects levels reported in the scientific literature.

Comparisons between CoPC concentrations in willow and birch leaves from coastal plain and tundra stations, and phytotoxicity thresholds for vascular plants reported in the literature, showed exceedances of the lowest aluminum and cadmium thresholds at 10-m stations and exceedances of the lowest zinc threshold at almost every site station and two reference stations (Table 6-17). Zinc concentrations also exceeded the highest minimum threshold at stations TT2-0010 and TT5-0010 near the port (Table 6-17). Cobalt concentrations also exceeded their lowest threshold, but site concentrations were within the range of reference concentrations (Table 6-17). Aluminum and cadmium exceedances for shrubs were limited to dusty areas near the road, and these results do not indicate potential for widespread toxicity to shrubs from exposures to these two metals. Zinc concentrations in shrub leaves tended to be elevated at the site relative to reference values, and concentrations at most site stations were greater than the normal zinc range reported for plant foliage (15–150 mg/kg dry weight; Langmuir et al. 2004). However, Nissen and Lepp (1997) reported mean zinc concentrations of approximately 80–300 mg/kg dry weight in washed leaves from eight *Salix* species, indicating that some willows can tolerate higher tissue concentrations. Zinc is an essential nutrient for plants and is not highly phytotoxic; however, symptoms of phytotoxicity such as leaf chlorosis and reduced plant growth are known to occur in contaminated soils, especially under acidic, oxidizing conditions where zinc is soluble and readily available to plants (Kabata-Pendias and Pendias 1992; McBride 1994). Based on the results of the phytotoxicity threshold comparisons for shrubs, zinc exposure in the DMTS road corridor cannot be ruled out as a potential contributing factor for observed effects to vascular plants.

Comparisons of CoPC concentrations in sedge blades from coastal plain and tundra stations with phytotoxicity thresholds showed that only aluminum and zinc concentrations at 10-m stations exceeded their lowest threshold, and no concentrations exceeded their highest reported threshold (Table 6-18; see Section 6.2.2). Aluminum and zinc concentrations in sedge blades were at most about 2-fold higher than their lowest phytotoxicity thresholds, and these exceedances occurred only at stations where heavy dust was visible on plant foliage, and where surface metals may account for a large fraction of the total metals concentrations measured in plant tissue samples. The results suggest a low potential for adverse effects to graminoid populations from CoPC exposures at the site.

Nonvascular plants seem to be more sensitive to metals than higher plants. In a field study of coniferous forest vegetation surrounding a brass foundry in Sweden, where copper and zinc concentrations in raw humus ranged from 20–8,400 and 90–6,300 mg/kg organic dry weight, respectively, Folkeson and Andersson-Bringmark (1988) reported that moss and lichen species richness and covers of dominant mosses and lichens declined significantly with increasing proximity to the foundry and increasing soil metals load, whereas no consistent relationship was found between the metals gradient and field-layer plants, such as grasses and shrubs. Feather mosses including *Pleurozium schreberi* and *Hylocomium splendens* appeared to be more sensitive to metals than the reindeer lichens *Cladonia arbuscula* and *C. rangiferina*, although dead individuals of all these species were observed near the foundry. Sulfur dioxide emissions, which often accompany metals pollution and can be toxic to vegetation, were not significant at this foundry site. However, the foundry's metal oxides emissions raised the surface soil pH up to 2.5 units higher than the normal ambient pH, a condition that may confound the relationships between observed vegetation effects and metals concentrations. Salemaa et al. (2001) observed a similar response in plant communities along a heavy metal and sulfur gradient near a copper-nickel smelter in Finland, where effects to understory vegetation were more pronounced than effects to the tree canopy, and common mosses such as *P. schreberi* appeared to be the most sensitive species, followed by lichens such as *Cladina* spp.

Based on the sensitivity ranges for moss reported by Folkeson and Andersson-Bringmark (1988), copper concentrations in moss near the port and along the DMTS road were below effects thresholds and therefore unlikely to cause adverse effects to moss. Zinc concentrations in moss were potentially high enough to cause mortality in mosses up to 100 m from the road and up to 1,000 m from port facilities, and to cause reductions in cover up to 1,000 m from the road (Table 6-19). Zinc concentrations in *Peltigera* and *Cladina* lichens at the port and along the road were potentially high enough at 10-m stations and some 100-m stations to result in moribund individuals or reductions in cover (Table 6-20). Zinc concentrations in lichens at 1,000-m stations along the road were below toxicity thresholds for lichens. At the port, lichen concentrations were above toxicity thresholds at station TT5-1000 (450 m from sources) but were below them at station TT5-2000 (1,430 m from sources; Table 6-20). However, this comparison is conservative in that the form of zinc in smelter emissions would be largely zinc oxide, whereas the predominant form in dust from Red Dog operations would be zinc sulfide, which may have a lesser effect on non-vascular plants than the more highly oxidized smelter emissions. Additionally, although the Folkeson and Andersson-Bringmark (1988) study had zinc concentrations comparable to those found in some areas near the DMTS, copper concentrations were much higher than are present at the DMTS, and thus the zinc thresholds may be conservative if copper contributed to toxicity in that study.

These comparisons with literature values suggest that zinc may be a contributing factor to the lower moss cover and lower lichen frequency and cover observed in tundra communities along the DMTS road (Figure 6-4). Seaward (1995) suggested that zinc, iron, and lead concentrations in water were responsible for creating a stress situation for lichen, primarily attributable to the zinc, but secondarily to iron and lead, which may play an independent or synergistic role. Similarly, Nash (1972) sampled lichens in the vicinity of an isolated zinc smelter in Pennsylvania that is a known source of sulfur dioxide and heavy metals, including cadmium and zinc. He suggested that lichen communities were markedly reduced and less diverse in the area known as the Lehigh Water Gap than in the nearby Delaware Water Gap control area. The heavy metal concentrations in lichens were up to four times as great as the soil values, and few species occurred in areas contaminated by zinc. Nash (1972) also examined sulfur dioxide concentrations and non-pollution factors, such as substrate, microclimate, and fire regimes. He concluded that zinc was the more important pollutant at this site because it was 100 times as high as the cadmium concentrations, and because sulfur dioxide was not detectable at the periphery of the lichen impoverished zone. Folkeson (1984) also reported that while some mosses and lichens are very sensitive to heavy metal pollution, there are a number of less common species that are tolerant and have healthy appearances even in nearby surrounding areas of brass mills in Sweden, where copper and zinc are the primary pollutants. Folkeson (1984) also reported that there is a tendency for tolerant species to increase in cover in an intermediate portion of the metals gradient because they are able to take advantage of diminishing competition from more sensitive species. Buck et al. (1999) also suggested that certain lichens are common colonists of metal enriched substrates, particularly those that are lead and zinc enriched. These species were found in areas of zinc-contaminated soils, but were not found in areas between contaminated soils. Richardson (1992) suggested that some lichens exhibit tolerance to metals either as a result of secreting oxalic acid, which forms insoluble metal oxalates or fungal strands that regulate the amounts of metals reaching the thallus from metal-contaminated soils. Richardson (1992) has stated that metals are usually in an insoluble form and cause little harm, but occasionally, accumulated dust and rock particulates do affect lichens, and can lead to the replacement of normal lichen flora with alternative lichen species. All of the studies discussed above indicate that the effects of heavy metals on lichens and mosses can vary depending on circumstances, and that individual studies are needed to determine what metal forms, concentrations, or physical factors are responsible for toxicity effects.

While Folkeson and Andersson-Bringmark (1988) and others related adverse effects in moss and lichen populations to copper and zinc concentrations, other CoPCs may be more phytotoxic. The relative toxicity of metals to lichens, for example, was reported in Tyler (1989) as follows: mercury, silver > copper, cadmium > zinc, nickel ≥ lead. In terms of absolute concentration, however, lead and zinc are typically one to two orders of magnitude higher than cadmium in lichen and moss samples from the site, and several orders of magnitude higher than mercury or silver. Adverse effects to lichen and moss communities are probably a result of simultaneous exposure to multiple stressors, including these metals. For example, in the Netherlands, concentrations of trace elements (including antimony, arsenic, bromine, cadmium, cesium, calcium, cerium, chromium, cobalt, iron, lanthanum, mercury, nickel, potassium, samarium, scandium, sodium, selenium, thorium, and zinc) were determined in epiphytic lichen species attached to the bark of trees, and concentrations of atmospheric trace gases were estimated at the sites of collection. Atmospheric sulfur dioxide and nitrogen dioxide were the most important factors determining lichen biodiversity, while effects of trace elements were very slight (Van Dobben et al. 2001).

As stated above, lichens are also known to be sensitive to sulfur dioxide (Nash and Gries 2002; Richardson and Nieboer 1983; Belandria 1989; see Section 6.6.3, *Uncertainties Related to CoPC Screening*). It is possible that sulfur dioxide emissions from vehicle traffic on the DMTS road and from power generation at the mine may make some contribution to the observed effects, but the degree to which that may be the case is unknown. Sulfur in the mineral concentrates is primarily in the form of sulfides (e.g., zinc sulfide and lead sulfide). Although this form of sulfur has a relatively low bioavailability, it is unknown at this time whether sulfur may have any contribution to effects observed in plant communities.

The DMTS road likely affects other environmental characteristics that were not assessed in this study, but which could exert considerable influences on the tundra landscape. For example, applications of the hygroscopic salt, calcium chloride, used to suppress road dust, likely increase the salinity of the adjacent tundra, which could result in brine effects such as a reduction in total plant cover or changes in community composition near the road (Walker 1996). Also, permanent roads in permafrost environments can impede natural drainage patterns and alter snowdrift (Walker 1996). Deeper snow accumulation near roads insulates the ground during the winter and reduces heat loss from the permafrost, and in the spring, road dust on the snow absorbs radiation, encouraging faster snowmelt. As a result, the seasonal thaw near roads may be deeper than in the surrounding tundra, which may hasten the onset of plant growth and reproduction, and may favor some species over others (Auerbach et al. 1997).

6.2.3.1.1 Conclusions

Reduced evergreen shrub, moss, and lichen covers and lower lichen frequency are the primary differences observed between tussock tundra plant communities near the DMTS road and comparable reference communities or less exposed site communities. As discussed previously, these plant types are among the most sensitive to dust deposition, elevated soil pH, and atmospheric deposition of CoPCs. Because all these factors are associated with fugitive dust from the road, they are interrelated, confounding their relationships with vegetation community characteristics. Most likely, a combination of physical and chemical road-related effects have altered the structure and possibly the function of the tussock tundra within 100 m of the DMTS road. At approximately 1,430 m from port facilities (station TT5-2000), evergreen shrub cover, moss cover, and lichen frequency were similar to coastal plain reference conditions. By 1,000 m from the road in the tundra community, moss cover had leveled off, and evergreen shrub cover and lichen frequency were comparable to reference levels. Thus, by 1,000 m from the road or 1,430 m from the port, coastal plain and tundra communities seem to recover from most road effects. However, lichen covers at 1,000-m and 2,000-m stations were significantly (2 to 4.5-fold) lower than reference covers, indicating that lichen effects are present at these distances from the DMTS road corridor. Further study would be required to define the full nature and extent of lichen effects related to fugitive dust deposition from the DMTS port, road, and Red Dog Mine, and to identify the causative agent(s) of lichen decline.

6.2.3.2 Supplemental Evaluation of Port Site Vegetation

Vegetation surveys conducted during the Phase II field program did not examine tundra at the DMTS port west of coastal plain transect TT5. However, areas of stressed tundra vegetation were observed near port facilities during the Phase I field investigation in 2003. A qualitative investigation of the most apparent affected areas was conducted at that time. Noted areas of stressed vegetation included an area near the northwest corner of CSB1 (Photograph 49), tundra at the end of the ion exchange treatment system overflow ditch (Photograph 50), and a triangular area of stressed vegetation located between the DMTS road and the southwest corner of CSB1 (Photograph 51). Most of the vegetation in these zones appeared to be dead (i.e., the vegetation was gray or brown in color, and brittle and dry to the touch; Photographs 52–55). In some cases, sedge tussocks were loosely attached to the substrate and easily uprooted (Photograph 56). Dead sphagnum moss was observed. The vegetation effects were most severe in the degraded tundra at the northwest corner of CSB1, where the loss of the live vegetation mat had lowered the ground surface elevation, making the area appear slightly sunken relative to the surrounding tundra. In this location, inorganic soil was noted within 2–3 in. of the tundra surface, and large rocks were exposed (Photograph 57). In places, live sedges and other rooted plants were interspersed among the dead tussocks and patches of bare ground (e.g., see Photograph 53). Comparable areas of exposed soil and rock and dead or stressed vegetation were not observed in the terrestrial reference area.

During the Phase I investigation, tundra soil and moss were sampled at 10-m, 100-m, and 1,000-m points along terrestrial transect TT1, which originated at the northwest corner of CSB1 and was oriented downwind (north) of the building. The 10-m station coincided with the area of stressed vegetation shown in Photographs 53–57; no living *H. splendens* moss was identified at the 10-m station, and thus only tundra soil metals data are available for that station. Tissue samples from higher plants were not collected during the Phase I program, nor were environmental samples collected from the other areas of stressed vegetation described above. However, a separate sampling program was conducted by Teck Cominco to more fully delineate the extent of deposition in the facility areas (Teck Cominco 2003a).

Metals concentrations were quite elevated in tundra soil from stations TT1-0010 and TT1-0100. Cadmium, lead, and zinc concentrations measured at these two stations were 71.2 and 67, 10,400 and 3,600, and 11,500 and 15,000 mg/kg, respectively. These values were several-fold higher than tundra soil concentrations at terrestrial transect stations TT5-0010 and TT2-0010 near the port, and an order of magnitude higher than concentrations at stations TT3-0010, TT6-0010, and TT8-0010 along the road. Metals levels in moss from TT1-0100 were also among the highest concentrations measured at the port and across the site. Maximum lead and zinc concentrations at the site were measured in moss from this station (1,720 and 8,120 mg/kg, respectively).

The elevated metals concentrations in tundra soil and moss tissue and the proximity of the 10-m and 100-m stations to the CSB suggest that fugitive concentrate is responsible for the stressed and dead vegetation observed directly downwind of CSB1. Historically, port workers would open the CSB door for ventilation, but this is no longer the practice, as dust control inside the building has been improved.

6.2.3.3 Hillslope Mesic Open Shrubland

In contrast with the coastal plain and tundra plant community results, site stations in the hillslope community as a group were not significantly different from the hillslope reference station in plant functional group covers or community indices (Table 6-3). Moss cover was higher at all hillslope site stations than at the reference stations; moss species' covers were not assessed during the field program, but *H. splendens* and other feather mosses appeared to dominate in the hillslope community. These species may respond differently to the road's influence than *Sphagnum* mosses common to the tussock tundra communities. Lichen cover was lower at site stations than reference stations in the hillslope community, and stations near the road had the lowest lichen covers (Figure 6-4). Site and reference comparisons should be evaluated with caution, however, given the uncertainties associated with the selection of this reference station (see Section 6.6.2).

Changes in plant community composition along hillslope transect TT6 may be related to topographical differences among stations. Differences in slope, aspect, and elevation among stations were more dramatic at this transect than on coastal plain and tundra transects, and it is plausible that these differences are reflected in the vegetation community, although plant data for just three stations along a topographically variable transect may not be representative of the hillslope community at large. Station TT6-0010 was on a northwest-facing, sloping bench above a creek drainage, and it was located at higher elevation than the next survey station on the transect, TT6-0100 (Photograph 15). Station TT6-1000 was situated on a south-facing slope and knoll located approximately 130 ft higher than TT6-0010. Plant community indices (species diversity, evenness, richness, and area richness) seemed to follow the topographical pattern, decreasing at lower elevation at station TT6-0100 and then increasing with elevation at station TT6-1000 (Table 6-15). Also, the underlying substrate at TT6-0100 could be alluvial materials with a different character than elsewhere on the transect, and therefore changes in the plant community along transect TT6 could also reflect substrate differences.

Environmental sampling results show that hillslope vegetation up to 1,000 m from the road is exposed to road dust. Tundra soil concentrations of many CoPCs were elevated over reference levels at all stations along transect TT6 (Table 6-16). Qualitative evaluations of vegetation were corroborative. Dust was detected by touch on plant foliage at 10-m and 100-m stations; blackening, bleaching, or drying was observed on foliose lichens and on crowberry, blueberry, and lingonberry shrubs at station TT6-0010, and some willows were partially defoliated at station TT6-0100. However, field notes indicate that the area experiences heavy wildlife use, and herbivory may be a contributing factor to the observed defoliation of shrubs. Several species, including bear, caribou, and moose, have been observed in the vicinity of transect TT6, and signs of wildlife use were noted in the field log. The relative contribution of herbivory to defoliation versus that from other causes could not be determined in the field. In addition, browning and bleaching of shrubs was recorded at the hillslope reference station, TS-REF-11, suggesting other possible causes, such as seasonal dryness.

Comparisons of shrub leaf CoPC concentrations with phytotoxicity thresholds for vascular plants showed that zinc concentrations in willow leaves from stations TT6-0010, TT6-0100, and TT6-2000 exceeded the lowest phytotoxicity threshold, but that no other chemicals in shrub leaves exceeded corresponding thresholds (Table 6-17). Zinc concentrations in willow leaves did not exceed the highest minimum phytotoxicity threshold, however, and concentrations at the 10-m and 2,000-m stations were within the normal range of zinc levels in plant foliage (15–150 mg/kg per Langmuir et al. 2004; Table 6-17). No CoPC concentrations in sedge blades exceeded the phytotoxicity thresholds (Table 6-18). The tissue concentration comparisons suggest that, in general, CoPC concentrations in the vascular plants tested were not high enough to result in adverse effects to plant populations.

Comparisons of zinc concentrations in hillslope moss samples with phytotoxicity thresholds reported by Folkeson and Andersson-Bringmark (1988) indicate that zinc concentrations up to 2,000 m from the road are high enough to be potentially toxic to moss (Table 6-19). However, the hillslope vegetation survey showed that moss cover was higher at site stations than at the reference area, so some types of moss are apparently abundant near the road. Zinc concentrations were below toxicity thresholds for lichens reported by Folkeson and Andersson-Bringmark (1988, Table 6-20). However, as discussed above in the risk characterization for tussock tundra communities, several CoPCs, including mercury, silver, copper, and cadmium, seem to be more toxic to lichens than zinc, and could potentially account for some of the depression in lichen cover seen near the road.

6.2.3.3.1 Conclusions

The small number of stations assessed in the hillslope community survey (transect TT6) makes it difficult to determine whether exposure to fugitive dust is adversely affecting the vegetation community along that transect. Differences in the hillslope vascular plant community with distance from the road may be related to environmental factors such as slope, aspect, and topography. Plant tissue comparisons with phytotoxicity thresholds suggest that elevated metals could affect mosses in the hillslope community, although the cover results for mosses as a group did not reveal phytotoxic effects associated with the road (Figure 6-4). Based on the plant community survey results, lichens appear to be the most sensitive group to road effects (Figure 6-4). Limited numbers of samples and topographic factors make interpretation of results difficult, and it is not clear whether CoPCs, environmental conditions, or a combination of factors are responsible for the observed differences in hillslope community stations. Further study would be required to define the full nature and extent of lichen effects related to fugitive dust deposition from the DMTS port, road, and Red Dog Mine, and to identify the causative agent(s) of lichen decline.

6.2.4 Risk Characterization for Tundra Soil Fauna

The structure and function of tundra soil fauna communities are not evaluated quantitatively in the ERA. 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 resulting from 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. Sampling conducted in 2004 indicated the presence of a diverse terrestrial invertebrate community at the site and reference locations. Figure 6-7 shows the composition of soil invertebrate samples collected in pitfall traps at site and reference stations. A photograph of a typical sample of invertebrates is included in Appendix J.

6.3 Freshwater Aquatic Life Assessment

Ecological receptors that inhabit freshwater aquatic environments along the DMTS road corridor include aquatic invertebrates, wetland and aquatic plants, and fish (Table 6-1). Risks to stream and tundra pond invertebrate communities, stream and tundra pond plant communities, and stream fish communities are evaluated in the following sections. Aquatic invertebrates were sampled in site and reference streams during the 2004 field program, and a comparative analysis of these invertebrate communities is presented in Section 6.3.1. Preliminary sampling as part of the Phase I investigation indicated that macroinvertebrates were uncommon in ponds, probably because of the nature of the substrate in these ponds, although this also could be because of the time of the season when sampling was conducted or the methods used to collect invertebrates. Risks to tundra pond invertebrates were not assessed directly through field investigations such as community surveys. However, a brief discussion of tundra pond invertebrate communities is presented in Section 6.3.2. Stream and tundra pond plant communities were assessed qualitatively through field observations, and CoPC concentrations in stream and pond plants were compared against phytotoxicity thresholds reported in the scientific literature. These results are reported in Section 6.3.3. Section 6.3.4 presents an exposure assessment and risk characterization for fish in streams that intersect the DMTS road. The fish assessment is based on two lines of evidence: site and reference comparisons of stream sediment and invertebrate CoPC concentrations, which represent potential exposures for fish; and the results of ongoing aquatic biomonitoring in the vicinity of the DMTS road and mine area. Birds and mammals that forage in freshwater aquatic environments, such as the common snipe and muskrat, are addressed in Section 6.5, *Wildlife Assessment*.

6.3.1 Stream Invertebrate Community Analysis

The aquatic macroinvertebrate assemblages found in the drift of three streams potentially affected by the DMTS transportation corridor (i.e., site streams) were compared with the assemblages found in two reference streams. The three site streams were Anxiety Ridge Creek, Omikviorok River, and Aufeis Creek, which cross the DMTS road at increasing distances from the mine (Figure 4-3). The stations sampled in the three site streams were designated as Stations ARC-R, OR-R, and AC-R, respectively.

In addition to the three site streams, three reference streams were sampled in a reference area located in the vicinity of Evaingiknuk Creek, approximately 20 km in the prevailing upwind direction from the Red Dog Mine (Figure 4-3). Because the three reference streams were unnamed, they were referred to as Reference Streams 3, 5, and 6 and the corresponding sampling stations were referred to as Stations ST-REF-3, ST-REF-5, and ST-REF-6, respectively. All three site streams drain to the northwest, crossing the DMTS road, whereas all three reference streams originate in the reference area and drain to the southeast. There is no evidence that any of the reference streams are affected by mine activities.

Although three reference streams were sampled, it was found during the sampling activities that the physical characteristics of Reference Stream 5 were not similar to those found at the three site streams or the other two reference streams (see discussion below in Section 6.3.1.4). That stream was therefore not used in the comparisons of macroinvertebrate assemblages between site and reference streams. Nevertheless, all of the information on the stream characteristics and macroinvertebrate assemblages collected in Reference Stream 5 during the present study is presented in this report for informational purposes.

Although not a part of the evaluation of macroinvertebrate drift assemblages, tissue concentrations of cadmium, lead, and zinc were measured in representative benthic macroinvertebrates collected from the bottom of each site stream and two reference streams (i.e., ST-REF-3 and ST-REF-6) using kick nets. Those results were used in the exposure characterization for stream fishes, and are discussed in detail in Section 6.3.4.1.

6.3.1.1 Field Sampling Methods

All stream invertebrate community sampling was conducted between June 22 and 25, 2004. All sampling methods were identical to those specified in the NPDES permit for the Red Dog Mine (DFG 1998b). For each stream, a single station was selected for analysis. The stream stations, which were adjacent to the DMTS road, covered a reach of stream typically 150–300 ft (45–90 m) in length from the edge of the road. For the three site streams, the stream segments were located immediately downstream from the DMTS Road. The segments in all three site streams were high-gradient environments with substrates composed primarily of cobble and gravel. The three reference streams were selected so that their physical characteristics were as similar as possible to those of the site streams.

Aquatic macroinvertebrates were sampled using drift nets set within riffle habitats. Five drift nets (mesh size = 363 microns) were installed as replicate samples at random locations within each stream segment. The dimensions of each drift net were 28 cm (11 in.) in height and 47 cm (18.5 in.) in width. In all cases, drift nets were deployed in water approximately 28 cm deep, so that the nets sampled the entire water column. Water velocity and depth were measured at the mouth of each drift net to determine the volume of water sampled, so that macroinvertebrate abundances could be standardized to water volume (i.e., m^3). Each drift net was deployed for a sampling period of approximately an hour (i.e., 55–83 minutes). Water velocity was measured at the mouth of each drift net using a flow meter at the beginning and end of each sampling period.

At the end of sampling, each drift net was removed from the stream and the retained material was rinsed into the screened end cup using site water, rinsing from the outside of the net. The screened end cup was then detached from the net, and the retained material was transferred to a sample container and preserved with a 10 percent formalin solution.

6.3.1.2 Laboratory Processing

In the taxonomic laboratory, each sample was transferred to a pan that was subdivided into quarters. All material was spread out in the pan and, in most cases, a one-quarter subsample was randomly selected for taxonomic analysis. In some cases, when macroinvertebrate densities were relatively low, taxonomic analysis was conducted on the entire sample.

All taxonomic determinations were made by a qualified taxonomic expert using a binocular microscope. Most insects were identified to the genus level, if possible. However, chironomids were identified only to the family level. All other taxonomic groups were identified to higher taxonomic levels (usually class or order).

Terrestrial invertebrates that were incidentally captured by the drift nets were identified to higher taxonomic levels and enumerated. However, the terrestrial invertebrates were not used in the comparisons of the site and reference streams.

6.3.1.3 Data Analysis

The results of the determinations of the taxa abundances in the macroinvertebrate drift assemblages from the various study sites were analyzed using a variety of benthic metrics that are commonly used to evaluate macroinvertebrates in stream environments. To be consistent with past studies conducted in the vicinity of the Red Dog Mine (e.g., Scannell and Ott 2001; Ott 2004), assemblages were characterized using the following four benthic metrics:

- **Total abundance:** total number of organisms per cubic meter of stream water, determined by calculating the volume of water that passed through each drift net (i.e., based on the duration of the sampling period, the mean stream flow, and the cross-sectional area of the water column sampled by the drift net)
- **Total taxa richness:** total number of unique taxa in each assemblage
- **Ephemeroptera, Plecoptera, and Trichoptera (EPT) relative abundance:** percentage of total abundance composed of three sensitive insect orders: Ephemeroptera, Plecoptera, and Trichoptera
- **Chironomid relative abundance:** percentage of total abundance composed of chironomids.

In addition to the four benthic metrics described above, two additional metrics recommended by Barbour et al. (1999) were also used to characterize the macroinvertebrate assemblages, including:

- **EPT taxa richness:** total number of unique taxa of three sensitive insect orders: Ephemeroptera, Plecoptera, and Trichoptera
- **Percent dominance:** percentage of total abundance composed of the most abundant taxon.

EPT taxa are commonly evaluated in assessments of freshwater benthic macroinvertebrate communities because these taxa are typically more sensitive than others to degraded water quality (U.S. EPA 1999d).

In addition to use of benthic metrics, a classification analysis was conducted using the Bray-Curtis similarity index applied to log-transformed abundances. The classification analysis is a multivariate technique that allows the various sampling stations to be compared based on the distribution of individuals among different taxa. Norris and George (1993) concluded that multivariate techniques show greater promise than univariate comparisons for detecting and understanding spatial and temporal trends of benthic macroinvertebrate communities.

6.3.1.4 Characteristics of Streams and Sampling Stations

All three site streams were similar with respect to size, water depth, stream velocity, and substrate composition, as shown in Photographs 58−62. Stream width at the drift-net placement locations ranged from approximately 5 to 15 m (15−50 ft), and maximum water depth at these placement locations was rarely greater than 30 cm (12 in.). The substrate of all three site streams was composed primarily of cobble and gravel.

The sampling characteristics for each drift-net sampling location are presented in Table 6-21, which contains information on water temperature, stream depth, stream velocity, sampling period, and sample volume. Mean velocity was similar among the three site streams, ranging from 5.1 ft/sec (1.6 m/sec) in the Omikviorok River to 5.7 ft/sec (1.7 m/sec) in Aufeis Creek. The total volume of water sampled for drift macroinvertebrates was also similar among the three site streams, ranging from 3,857 m^3 in the Omikviorok River to 4,700 m^3 in Aufeis Creek.

With respect to the three reference streams, Reference Streams 3 and 6 were considered similar to the three site streams with respect to size and substrate character (Photographs 63 and 64). By contrast, Stream 5 was generally narrower than the site streams and the substrate was primarily soft sediment (i.e., silt and sand), rather than cobble and gravel (Photographs 65 and 66). Mean stream velocity in Reference Stream 6 (i.e., 5.3 ft/sec, 1.6 m/sec) was within the range found for the site streams (Table 6-21). However, mean stream velocities in Reference Streams 3 and 5 (i.e., 2.9 and 2.6 ft/sec, respectively) were noticeably less than the range found for the site streams.

Because of the dissimilarity in substrate characteristics between the site streams and Reference Stream 5, the latter was considered inappropriate for use as a reference stream. Although mean stream velocity in Reference Stream 5 was noticeably less than the flows observed in the three site streams, it was retained as a reference stream because its size and substrate character were similar to those characteristics for the three site streams.

6.3.1.5 Comparisons of Macroinvertebrate Drift Assemblages

In this section, the results of the comparisons of macroinvertebrate drift assemblages between site and reference stations are described on the basis of benthic metrics and classification analysis.

6.3.1.5.1 Benthic Metrics

Abundances of the various benthic taxa found in the drift assemblages at the site and reference stations are presented in Table 6-22, and the corresponding benthic metrics for each station are presented in Table 6-23. The taxa abundances found at each of the five drift-net sampling locations within each station are tabulated in Appendix G. As expected, the drift assemblages at all sampling stations were dominated numerically by insects, including beetles (Coleoptera), springtails (Collembola), true flies (Diptera), mayflies (Ephemeroptera), stoneflies (Plecoptera), and caddis flies (Trichoptera). The true flies included the most families (i.e., 8), although they were dominated numerically at all stations by a single family (i.e., Chironomidae).

As described previously, six benthic metrics were used to compare macroinvertebrate drift assemblages between site and reference stations. The results of the comparisons based on each benthic metric were as follows:

- **Total abundance:** Values for this metric ranged from 6.6 individuals/m^3 at ST-REF-3 to 28 individuals/ m^3 at Station AC-R (Figure 6-8). Values of total abundance at all three site stations (10–28 individuals/ m^3) were within or slightly above the reference range of 6.6–24 individual/ m^3, indicating that no major differences between site and reference conditions were found for this metric.

- **Percent dominance:** This metric ranged from 37 percent at Station ARC-R to 63 percent at Station OR-R (Figure 6-8). Values of percent dominance at Stations OR-R and AC-R (60–63 percent) were within or slightly above the reference range of 50–61 percent, indicating that no major differences between site and reference conditions were found at those two site stations for this benthic metric. By contrast, percent dominance at Station ARC-R was considerably less than the reference range, indicating that the drift community at that station was more balanced than reference conditions. Chironomidae was the dominant taxon at Stations ARC-R, OR-R and ST-REF-6, whereas the mayfly *Baetis* was the dominant taxon at Stations AC-R and ST-REF-3.

- **Total taxa richness:** This metric ranged from 23 taxa at Station OR-R to 31 taxa at ST-REF-6 (Figure 6-9). Values of total taxa richness at all three site stations (i.e., 23–26 taxa) were slightly lower than the reference range of 27–31 taxa, indicating that the drift assemblages at the three site stations were slightly less diverse than the assemblages at the reference stations.

- **EPT taxa richness:** This metric ranged from 9 taxa at Stations ARC-R and OR-R to 15 taxa at Station ST-REF-5 (Figure 6-9). Values of EPT taxa richness at all three site stations (9–10 taxa) were less than the reference range of 14–15 taxa, indicating that the EPT components of the drift assemblages at the three site stations were less diverse than the EPT components of the reference assemblages.

- **EPT relative abundance:** This metric ranged from 15 percent at Station OR-R to 82 percent at Station AC-R (Figure 6-10). EPT relative abundances at Stations ARC-R and AC-R (49–82 percent) were within or greater than the reference range of 29–71 percent. By contrast, the value found at Station OR-R was approximately half the minimum reference value, indicating that the contribution of EPT taxa to the total drift assemblage at this station was noticeably less that the contribution found for all other stations, both site and reference.

- **Chironomid relative abundance:** This metric ranged from 13 percent at Station AC-R to 63 percent at Station OR-R (Figure 6-10). Chironomid relative abundances at all three site stations (13–63 percent) were within or only slightly outside the reference range of 17–61 percent, indicating that no major differences between site and reference conditions were found for this metric.

In summary, the comparisons of benthic metrics between site and reference stations indicated that for most metrics, the site stations were similar to or only slightly different from the reference stations. The only exceptions were for EPT taxa richness, which was lower than the reference range at all three site stations, and for EPT relative abundance, which was lower than the reference range at Station OR-R.

6.3.1.5.2 Classification Analysis

The results of the classification analysis are presented as a dendrogram in Figure 6-11. In general, the results indicate that the assemblages at the site stations were similar to those at the reference stations, as no major outliers were identified on the dendrogram. Station ARC-R exhibited the highest similarity with a reference station (i.e., ST-REF-3). Station ST-REF-6 then clustered with those two stations, followed by Stations OR-R and AC-R.

6.3.1.6 Risk Characterization for Stream Invertebrate Communities

The results of the evaluation of benthic macroinvertebrate drift assemblages at the three site streams and two reference streams showed that all assemblages comprised a relatively large number of taxa and that, overall, assemblages in the site streams were similar to those in the reference streams.

Although only 9–10 EPT taxa were found in the drift assemblages in the three site streams, EPT taxa comprised relatively large proportions of the assemblages (49–82 percent) at Stations ARC-R and AC-R. At Station AC-R, total abundance was the highest observed in the study and chironomid relative abundance was the lowest value observed. Although percent dominance and chironomid relative abundance were elevated at Station OR-R, they were comparable to the values observed at Station ST-REF-6. These additional characteristics of the drift assemblages found in the three site streams, combined with the relatively large numbers of total taxa found in those assemblages (23–26 taxa) and the results of the classification analysis, indicate that the overall characteristics of the assemblages found in the three site stream stations were similar to reference conditions.

As described above, tissue concentrations of cadmium, lead, and zinc in representative benthic macroinvertebrates were measured in each site stream and in two reference streams, and the results are discussed in detail in Section 6.3.4.1. Tissue concentrations of all three metals in macroinvertebrates from Aufeis Creek and Omikviorok River were within the concentration ranges found in the two reference streams. Cadmium and lead concentrations in samples from Anxiety Ridge Creek were slightly elevated over reference concentrations, but zinc concentrations were within the reference range. These benthic macroinvertebrate tissue results generally agree with the results of the evaluations of macroinvertebrate drift assemblages in that no large differences were found between the site and reference streams.

6.3.2 Tundra Pond Invertebrate Assessment

Tundra pond invertebrate communities were not sampled during the 2004 supplemental sampling program. Sediment sampling conducted as part of the Phase I investigation suggested that macroinvertebrates were uncommon in ponds, as none were seen in the grab samples at site or reference stations. Some ponds, such as TP1-0100, were merely low-lying areas of flooded tundra that may not support a resident benthic community (Photograph 4). Sediment samples collected from ponds such as TP1-0100 were composed primarily of matted vegetation, which may not provide appropriate habitat for macroinvertebrates. In addition, small ponds in flooded tundra may be ephemeral, shrinking or vanishing during dry periods. For these reasons, it is unlikely that there are substantial macroinvertebrate communities in some small tundra ponds. However, if macroinvertebrates are present in larger, permanent tundra ponds (for example, see Photograph 5), the potential for adverse effects to these receptors can be assessed indirectly based on the results of the coastal lagoon sediment toxicity tests, described in Section 6.4.1. Test results indicated that site lagoon sediments were no more toxic to the test organism (amphipods) than control sediments, and survival rates were high for all coastal lagoon sediments. Comparison of CoPC concentrations in tundra pond sediments with concentrations in lagoon sediments used in toxicity testing can therefore be used as a means of assessing whether toxicity to macroinvertebrates would be expected in tundra ponds. Sediments were sampled from four tundra ponds at the site (TP1-0100, TP1-1000, TP2-0100, and TP2-1000) during the Phase I program in 2003, and CoPC concentrations in pond sediments are reported in Appendix C. Lagoon sediment data from 2003 and 2004 are available in Appendices C and G, respectively.

Four chemicals (cadmium, lead, mercury, and zinc) were retained as CoPCs in tundra pond sediment in the ecological screening based on comparisons against screening benchmarks and reference sediment concentrations (Sections 3.5 and 3.6; Table 3-38). An additional seven chemicals (antimony, cobalt, fluoride, molybdenum, silver, strontium, and tin) did not have relevant screening benchmarks and could not be eliminated based on reference comparisons. Of these CoPCs for pond sediment, cadmium, lead, and zinc were also analyzed in coastal lagoon sediments associated with the toxicity tests, and concentration ranges for these metals at site lagoon stations were 0.31–15.8 mg/kg, 14.9–481 mg/kg, and 103–3,210 mg/kg, respectively. Maximum cadmium, lead, and zinc concentrations in tundra pond sediments were 119 mg/kg, 2,180 mg/kg, and 27,000 mg/kg, respectively (Table 3-22). These maximum concentrations were measured in sediments from pond station TP1-0100, located near facilities at the port. However, concentrations of all three metals in sediments at the other three tundra pond stations at the site were approximately an order of magnitude lower, on average, than the maximum lagoon sediment concentrations measured in lagoon samples. Thus, tundra ponds located along the central portion of the DMTS road, such as TP2-0100 and TP2-1000, and ponds located farther from port facilities, such as TP1-1000, had cadmium, lead, and zinc concentrations in sediment that were well below the maximum no-effects concentrations determined for lagoon invertebrates through toxicity testing. These results suggest that the likelihood of adverse effects to macroinvertebrates from exposure to CoPCs in ponds along the road is low. Tundra pond TP1-0100 appears to be influenced by local sources of CoPCs at the port, as is the surrounding tundra, and its sediment concentrations exceeded the no-effects concentrations for lagoon invertebrates. The lowest effect concentrations for cadmium, lead, and zinc are not known based on the results of the toxicity tests, as exposure to site sediments did not reduce amphipod survival rates compared to controls. Therefore, the likelihood of adverse effects to macroinvertebrates in ephemeral tundra ponds near port facilities (such as TP1-0100) cannot be evaluated.

6.3.3 Freshwater Aquatic Plant Community Assessment

Freshwater plant communities in the DMTS road corridor include stream and tundra pond vegetation. Vegetation at stream sampling stations consisted primarily of a tall willow community with understory plants lining the stream banks, and forbs and graminoids growing on gravel bars in the stream channel (Photographs 1 and 2). Site and reference streams tended to be fast-moving watercourses with cobble and gravel substrates, as described in Section 6.3.1, and plants were generally rooted at the stream margins or on high spots and did not line the stream channel. Some algae were observed growing on rocks in streambeds. Tundra pond vegetation was typically a fringe of hydrophytic sedges such as *C. aquatilis* and *E. angustifolium* that transitioned into tussock tundra away from the pond (Photograph 42). Floating aquatic macrophytes were observed in some of the larger ponds (Photograph 5).

Risks to stream and pond vegetation from exposure to CoPCs are assessed in the sections below. Freshwater plant communities were evaluated qualitatively through field observations, but community structure was not assessed quantitatively through vegetation surveys, such as those conducted for the terrestrial and lagoon plant community assessments (see Section 6.2.1). Qualitative assessments of plant vitality are summarized below. The CoPC concentrations in plant tissues (whole sedge plants and willow leaves) collected from streams and ponds were compared against phytotoxicity thresholds reported in the scientific literature. The results of these comparisons are also presented below. Field observations and phytotoxicity threshold comparisons are integrated in the risk characterization for stream and pond plant communities (Section 6.3.3.3).

6.3.3.1 Field Observations

Stream vegetation was observed over the length of the stream stations, which were adjacent to the DMTS road, and covered a reach of stream typically ranging from 150 to 300 ft from the edge of the road. The vegetation at site stations was notably dusty, especially at the portion of the stream reach near the DMTS road; however, no signs of phytotoxicity were observed. In Aufeis Creek, willow leaves were visibly dusty at the start of the sampling reach (located near the bridge crossing the creek) and were slightly dusty to the touch by the end of the sampling reach (approximately 150–300 ft downstream). In Anxiety Ridge Creek and the Omikviorok River, heavy dust was visible on willow leaves near the road, and dust on leaf surfaces decreased away from the road. Sedge plants growing near the road in the Omikviorok River had dusty or sandy inflorescences. No dusty foliage or signs of phytotoxicity were reported at reference streams.

Tundra pond sedges were not visibly dusty at site or reference stations. Sedges at site station TP1-0100 and reference station TP-REF-3 had blunt tips, suggesting that they had been grazed. Some sedge plants at TP1-0100 had sticky seed heads, and a type of insect or mite was observed under the blades of these plants. No obvious signs of phytotoxicity in tundra pond plants were observed. Photograph 67 shows a tundra pond plant community at station TP-3, located near the middle portion of the DMTS road (location shown on Figure 4-3).

6.3.3.2 Comparisons with Phytotoxicity Thresholds

Table 6-24 summarizes the comparisons of whole sedge (*Carex aquatilis*) and willow leaf (*Salix planifolia*) samples collected from three site streams (Aufeis Creek, Anxiety Ridge Creek, and Omikviorok River) and three reference streams (Figure 4-3) against phytotoxicity thresholds for vascular plants. Sedge samples were whole plants collected along a 150–300 ft reach directly downstream of the road crossing. The shoots were unwashed, and the roots were rinsed in site water to remove external sediment particles. Dead material was plucked off the sedge samples to the extent possible in the field. Willow leaf samples were unwashed composites of leaves collected from approximately five shrubs along the length of the sampling reach. Because tissue samples were not washed before analysis, plant tissue CoPC concentrations include both metals inside plant tissues and metals in dust on plant surfaces. Therefore, concentrations likely overestimate the amount of CoPCs actually taken up by the plants. Phytotoxicity thresholds are based on foliage concentrations, so the whole sedge samples, which contained root material, are not directly comparable to the threshold values. However, samples of sedge blades only were not collected in freshwater aquatic environments, so whole plant results were used in the comparison.

Aluminum and chromium concentrations in stream sedge samples exceeded phytotoxicity thresholds (Table 6-24). Both sedge samples from site streams exceeded the maximum phytotoxicity threshold for aluminum. However, all reference sedge samples also exceeded the maximum threshold (Table 6-24), suggesting that the range of aluminum threshold values for plant foliage may be overly conservative for entire sedge plants. Plant species, and even varieties within species, can differ greatly in their abilities to take up aluminum and transport it to shoots and leaves (Kabata-Pendias and Pendias 1992). The aluminum concentration in the sedge sample from the Omikviorok River exceeded the maximum threshold value by almost an order of magnitude (Table 6-24). Sediment particles that may have adhered to the sedge roots could be responsible for the high aluminum concentration in this sample, as the sediment concentration was several-fold higher than the plant tissue concentration (9,520 mg/kg compared to 1,900 mg/kg). The chromium concentration in this sample was also elevated relative to other site and reference samples and exceeded phytotoxicity thresholds (Table 6-24). No other chemicals exceeded phytotoxicity thresholds in stream sedge samples (Table 6-24).

Willow samples at stations AC-R (Aufeis Creek), ARC-R (Anxiety Ridge Creek) and OR-R (Omikviorok River) exceeded the minimum threshold for aluminum, and samples from AC-R and ARC-R also exceeded phytotoxicity thresholds for zinc (Table 6-24). Aluminum concentrations in willow samples from site streams were about 3-fold higher than the lowest phytotoxicity threshold but were below the maximum threshold value (Table 6-24). Zinc concentrations in site willows were about 2-fold higher than the lowest threshold and were below the highest minimum threshold (Table 6-24). No other chemical concentrations in site willow samples exceeded phytotoxicity thresholds. Reference willow samples did not exceed phytotoxicity thresholds for any CoPC (Table 6-24).

Whole sedge samples collected from four site tundra ponds (TP1-0100, TP1-1000, TP3, and TP4) and three reference ponds (Figure 4-3) were also compared to phytotoxicity thresholds, and the results are summarized in Table 6-25. Sedge concentrations exceeded aluminum, chromium, cobalt, lead, and zinc phytotoxicity thresholds at one or more tundra ponds at the site (Table 6-25). Reference sedge concentrations exceeded aluminum, arsenic, chromium, cobalt, and vanadium thresholds (Table 6-25). Reference ponds had sedges with the highest aluminum and chromium concentrations, which were several-fold higher than site sedge concentrations; therefore, exposures to aluminum and chromium in site tundra ponds do not represent incremental risks to freshwater plants. Cobalt concentrations exceeded the lowest threshold at one site station (TP1-1000) and one reference station (TP-REF-5), but the concentrations did not exceed the highest minimum threshold (Table 6-25). Lead concentrations exceeded the lowest phytotoxicity threshold at two site ponds, but the exceedances were relatively low (21.1 and 48.1 mg/kg, as compared to a threshold range of 20–300 mg/kg). Zinc concentrations also exceeded the lowest threshold value in these two site ponds but did not exceed the highest minimum threshold for zinc (Table 6-25). Sedges with the highest lead and zinc concentrations were collected at Station TP1-0100, a flooded depression in the tundra located near the concentrate conveyor and other port facilities (Table 6-25; Photograph 4).

6.3.3.3 Risk Characterization for Freshwater Aquatic Plants

Stream vegetation communities near the DMTS road experience higher exposures to some CoPCs, such as aluminum and zinc, than reference communities, as reflected by their respective tissue concentrations (Table 6-24). Plants at site stream stations were visibly dusty, particularly near the road, where they may have been within range of road splatter. Foliage dust levels diminished with increasing distance from the road. Therefore, plant samples collected at site stream stations represented the greatest exposure scenarios for stream vegetation. Even so, only two chemicals in sedges (aluminum and chromium) and two chemicals in willow leaves (aluminum and zinc) exceeded phytotoxicity thresholds (Table 6-24). Aluminum exceedances for sedge samples may be related to differences between foliage and whole plant concentrations, to the presence of external dust on blades, or possibly to residual sediment on roots. If sediment particles from the Omikviorok River were adhered to the corresponding sedge sample, for example, the aluminum concentration in the sediment (9,520 mg/kg) could account for the elevated concentration measured in the sedge sample (1,900 m/kg). Sedges at the site and reference stations had a similar appearance and no obvious indications of phytotoxicity. Metal-tolerant higher plant species tend to belong to a few families, including Cyperaceae, the sedge family (Kabata-Pendias and Pendias 1992). Thus, sedge tissue concentrations of CoPCs in excess of generic phytotoxicity thresholds do not necessarily indicate a potential for adverse effects to this plant group. Concentrations of aluminum and zinc in willow leaves were elevated over reference concentrations in most cases, but willow shrubs growing along site stream banks were tall, robust shrubs that did not appear to be affected by the road, except for foliage dust levels (Photographs 61 and 62). Therefore, there is low likelihood that stream plant communities as a whole are adversely affected by exposure to site-related CoPCs.

In the tundra pond environment, sedges around site and reference tundra ponds seemed to be healthy, and dust was not detectable on their foliage. In site ponds, only cobalt, lead, and zinc concentrations in whole sedge plants exceeded phytotoxicity thresholds for plant foliage and representative reference concentrations (Table 6-25). Only one site sample had a cobalt concentration in excess of the lowest threshold value, and this CoPC also exceeded the lowest threshold at reference station TP-REF-5. Thus, elevated cobalt concentrations in sedges appear to be localized occurrences in both site and reference pond communities (Table 6-25). Lead and zinc concentrations in sedges were scarcely elevated above phytotoxicity thresholds at pond station TP4, although tissue concentrations were greater than the range of reference concentrations (Table 6-25). Lead and zinc concentrations were somewhat higher in sedges at TP1-0100, where plants are subject to dust deposition from port facilities. Based on qualitative observations made during field sampling, tundra pond plant communities located more than 100 m from the DMTS road do not appear to be adversely affected by fugitive dust. The results of the tissue comparisons with phytotoxicity thresholds and reference data also suggest low likelihood of risk to these pond plant communities, with the possible exception of ponds in low-lying areas to the southwest of the mine's ambient air/solid waste permit boundary (e.g., TP4), where incremental exposure to lead and zinc may occur. Note that ponds were not observed in the mountainous terrain surrounding the mine. Exceedances of phytotoxicity thresholds and reference tissue concentrations at pond TP-0100 indicate that adverse effects from lead and zinc are possible at ponds located near port facilities.

6.3.4 Stream Fish Assessment

Fish that inhabit streams that cross the DMTS road may be exposed to CoPC concentrations in water, sediment, and food items. Although the results of the surface water screening indicate a low likelihood of adverse effects to aquatic life, the sediment screening against toxicity benchmarks and reference concentrations retained several chemicals for further evaluation in the ERA (see Section 3.6.2.2). In the following sections, fish exposures to CoPCs in sediment and food items are evaluated with the results of aquatic biomonitoring studies used to assess the potential for adverse effects to stream fish populations.

6.3.4.1 Exposure Characterization

Fish were not sampled for metals analysis in site or reference streams in the 2004 supplemental sampling program. However, incremental exposure to CoPCs in site streams may be evaluated by comparing site and reference metals concentrations in stream sediments and fish prey (benthic invertebrates). Table 6-26 provides the cadmium, lead, and zinc concentrations in surface sediment and corresponding invertebrate tissue at two reference stations and stations at Aufeis Creek, Anxiety Ridge Creek, and Omikviorok River. Cadmium, lead, and zinc concentrations in sediment were greater at the site streams than at the two reference stations (Table 6-26). Metals concentrations in site streams were highest in Anxiety Ridge Creek (1.06, 117, and 148 mg/kg for cadmium, lead, and zinc, respectively), followed by samples in Aufeis

Creek (nearer to the port); concentrations from Omikviorok River (crossing the middle of the haul road) were the lowest.

Metals concentrations in crane fly larvae samples from Aufeis Creek and Omikviorok River were within the concentration ranges in invertebrate composites at the two reference stations (Table 6-26). Invertebrate samples from Anxiety Ridge Creek had the maximum cadmium, lead, and zinc concentrations in prey tissue (0.803, 10.9, and 96.2 mg/kg, respectively). Cadmium and lead concentrations in samples from Anxiety Ridge Creek were slightly elevated over reference concentrations (0.347–0.696 and 2.73–8.14 mg/kg, respectively), but zinc concentrations were within the reference range (91.3–137 mg/kg). In the three site streams, invertebrates from Aufeis Creek had the lowest cadmium and lead concentrations, and invertebrates from Omikviorok River had the lowest zinc concentration (Table 6-26).

In addition to cadmium, lead, and zinc, two other metals (arsenic and nickel) were retained as CoPCs in stream sediment based on exceedances of screening benchmarks and reference concentrations. Additional chemicals could not be eliminated based on reference comparisons, and in the absence of relevant screening benchmarks, they were also retained as CoPCs (including antimony, cobalt, fluoride, molybdenum, selenium, silver, strontium, thallium, and tin). These other metals may be elevated at road stations in site streams relative to reference concentrations; however, sediment concentrations near the road may be higher than concentrations elsewhere in the stream habitats (Appendix C).

6.3.4.2 Aquatic Biomonitoring Results

In 2002, DFG conducted a study of metals concentrations in juvenile Dolly Varden from creeks that cross the DMTS road, including Aufeis Creek, Omikviorok River, and Anxiety Ridge Creek (Ott and Morris 2004). Fish were sampled at stations located upstream and downstream of the DMTS road in order to investigate the road's potential effect on fish tissue metals concentrations. Creeks near the mine (Buddy Creek, North Fork Red Dog Creek, Grayling Junior Creek, and Mainstem Red Dog Creek) were also sampled. The residence time of juvenile Dolly Varden in these creeks in not known, but for most sites, fish depart from their rearing areas in the fall to over-winter in lower reaches of the drainages and return to the sites in mid-June (Ott and Morris 2004).

Among creeks that cross the DMTS road, cadmium and lead concentrations were highest in fish from Anxiety Ridge Creek (Ott and Morris 2004). When the authors compared fish tissue concentrations among streams, they rated cadmium and lead as "low" in Aufeis Creek and Omikviorok River and "medium" or "high" in streams near the mine (Ott and Morris 2004). Selenium concentrations were rated "medium" in all streams except Mainstem Red Dog Creek, where levels were considered "high." Zinc concentrations were rated "low" in Aufeis Creek, "medium" in Omikviorok River, Anxiety Ridge Creek, Buddy Creek, and North Fork Red Dog Creek, and "high" in Mainstem Red Dog Creek and Grayling Junior Creek. Ott and Morris (2004) refer to low, medium, and high data ranges for cadmium of 0.03 to 0.21, 0.44 to 0.47, and 0.80 to 3.13 mg/kg, respectively. For lead, low, medium, and high referred to data ranges of 0.02 to 0.18, 0.25 to 0.73, and 8.4, respectively. For selenium, low, medium, and high referred to data ranges of 1, 2.2 to 7.2, and 12.7, respectively. For zinc, low, medium, and high referred to data ranges of 78.6 to 90.4, 111 to 124, and 170 to 286, respectively. It should be noted that in Red Dog Creek, metals concentrations have actually declined from historic levels as a result of the Red Dog Creek diversions, which has reduced water contact with mineral deposits.

Selenium tended to be higher in fish from North Fork Aufeis Creek (upstream of the road), Aufeis Creek (downstream of the road), and Anxiety Ridge Creek (upstream and downstream of the road) than in fish from the Omikviorok River. Zinc concentrations were similar in fish from Anxiety Ridge Creek and the Omikviorok River but were significantly lower in fish from Aufeis Creek. Anxiety Ridge Creek traverses terrain that is more mineralized than other creeks that cross the DMTS road (Kulas 2005, pers. comm.). Fish in Anxiety Ridge Creek may be accumulating metals from local lead and zinc mineralization 1 mile west of the DMTS road crossing (Grizzly Showing). Stream sediment samples collected in the 1970s (before mining operations) contained up to 253 mg/kg zinc upstream from the current Anxiety Ridge Creek bridge (Clark 2005, pers. comm.).

Cadmium and lead were significantly higher in downstream fish than upstream fish in Anxiety Ridge Creek (Ott and Morris 2004). Cadmium was also higher in downstream fish than upstream fish in Aufeis Creek. Selenium and zinc concentrations in fish from creeks that cross the DMTS road were not consistently related to capture location relative to the road (Ott and Morris 2004). A multiyear comparison of Dolly Varden samples from Anxiety Ridge Creek, Mainstem Red Dog, and North Fork Red Dog creeks indicated that, in general, cadmium, lead, and selenium concentrations in juvenile Dolly Varden are not changing with time. A comparison of zinc concentrations between 2001 (when zinc was initially analyzed) and 2002 indicated that the median zinc concentrations in fish did not change significantly between the years.

Based on the study results, the authors recommended continued monitoring of metals concentrations in juvenile fish in creeks near the mine, including Anxiety Ridge Creek, but they suggested that further study of juvenile fish concentrations in Aufeis Creek and the Omikviorok River is not warranted, given the relatively low metals concentrations in fish from these creeks compared to streams near the mine, and the lack of clear evidence of a road effect on fish metals concentrations (Ott and Morris 2004).

Maximum whole body fish tissues in Anxiety Ridge Creek were similar to or lower than those found in Grayling Junior Creek, a tributary to the naturally mineralized Ikalukrok Creek, located north of the Red Dog Mine (Scannell and Ott 2006). Specifically, the maximum cadmium and zinc concentrations in fish collected from Anxiety Ridge Creek stations (upstream, downstream, and at the DMTS road) were 1.32 and 140 mg/kg (dry weight), respectively, as compared to cadmium and zinc concentrations in Ikalukrok fish (3.78 and 573 mg/kg, dry weight, respectively). Maximum lead (2.86 mg/kg, dry weight) and selenium (8.5 mg/kg, dry weight) concentrations in Anxiety Ridge Creek fish were similar to lead (1.44 mg/kg dry weight) and selenium (7.5 mg/kg dry weight) concentrations measured in Grayling Junior Creek fish.

6.3.4.3 Fish Tissue Comparisons with Effects Thresholds

Because significant differences were found between cadmium and lead concentrations in fish collected by Ott and Morris (2004) upstream and those collected downstream of the DMTS road in Anxiety Ridge Creek, chemical concentrations in juvenile Dolly Varden from Anxiety Ridge Creek were compared with critical tissue concentrations for freshwater fish as compiled by Jarvinen and Ankley (1999, Table 6-27) as a method of screening to see if these tissue levels may indicate the possibility of adverse effects.

Dolly Varden tissue residue data were compared against no-effect and lowest-adverse effect levels for ecologically relevant endpoints, including survival, growth, and reproduction. Maximum concentrations of cadmium (0.308 mg/kg), lead (0.612 mg/kg), and selenium (2.01 mg/kg) in fish collected near or downstream of the DMTS road were greater than the lowest reported effects thresholds, but were also within the ranges of reported no-effects levels. Maximum cadmium and selenium concentrations in fish collected upstream of the road also exceeded the lowest effect threshold. The maximum zinc concentration in fish tissue (36.1 mg/kg) was below the lowest threshold for effects. Thus, based on a direct comparison to critical tissue residue levels developed in some freshwater fish studies, cadmium, lead, and selenium concentrations in some juvenile Dolly Varden were high enough to suggest a potential for adverse effects. However, because measured tissue concentrations of these metals (with the exception of selenium, which exceeds the freshwater salmonid no effects threshold) are also below the maximum NECs, adverse effects to fish cannot be conclusively predicted, as the sensitivity of Dolly Varden relative to the test species is not known.

6.3.4.4 Risk Characterization for Freshwater Fish

Based on the results of site and reference comparisons in Section 6.3.4.1, fish in streams that cross the DMTS road may have incremental exposures to CoPCs in sediments, but not significantly different exposures in prey, since invertebrate concentrations were generally comparable in site and reference streams (Table 6-26). Fish in Anxiety Ridge Creek may be exposed to slightly higher cadmium and lead concentrations than fish in reference creeks (Table 6-26). However, because prey concentrations are generally comparable between site and reference, incremental exposure to CoPCs may not be pronounced for fish feeding in creeks that cross the road.

Juvenile Dolly Varden captured downstream of the road in Anxiety Ridge Creek had elevated cadmium and lead levels relative to fish captured upstream of the road, perhaps reflecting a road effect on sediment metals concentrations in this creek (Ott and Morris 2004). Fish metals concentrations in Aufeis Creek and the Omikviorok River did not show a consistent pattern related to proximity to the road. Based on comparisons with critical tissue concentrations in freshwater fish, adverse effects to individuals from cadmium and selenium exposures are possible both upstream and downstream of the road in Anxiety Ridge Creek, and adverse effects from lead exposure are possible downstream of the road (Table 6-27). However, these comparisons do not necessarily suggest a likelihood of unacceptable risk to fish, because ranges of no-effects and effects concentrations overlap considerably, as shown in Table 6-27. Incremental exposure to CoPCs in sediment does not appear to translate into population-level effects in site creeks. Ott and Morris (2004) suggested that the juvenile Dolly Varden populations in creeks near the DMTS appear to be healthy, and that annual population fluctuations are attributable to environmental conditions. Overall, these findings indicate that risk from exposure to CoPCs is unlikely to have an adverse effect on the abundance of fish in streams that cross the road.

6.4 Coastal Lagoon Aquatic Life Assessment

Risks to coastal lagoon aquatic life, including benthic invertebrates and wetland plant communities, are evaluated in the subsections below. Benthic invertebrates may be exposed to site-related CoPCs in sediment, and therefore toxicity tests were conducted on sediments from the Port Lagoon North and the North Lagoon, located in the vicinity of the DMTS port, and on sediments from two reference lagoons located south (prevailing upwind direction) of the DMTS port (Figure 4-4). The toxicity test results are summarized in Section 6.4.1. To assess potential effects to coastal lagoon vegetation, plant community characteristics such as species composition, dominance, and richness were measured in wetland communities growing adjacent to these same lagoons, at the stations illustrated in Figure 4-4. In addition, CoPC concentrations in lagoon sedge samples were compared against phytotoxicity thresholds reported in the scientific literature. Risks to coastal lagoon plant communities are addressed in Section 6.4.2.

Port Lagoon North and the North Lagoon do not appear to have resident fish populations that would be exposed to CoPCs. Fish sampling was attempted in site and reference lagoons during the 2004 supplemental sampling program, using baited minnow traps and beach seines, but no fish were caught in any lagoon. In the pre-mine baseline studies (Dames & Moore 1983a,b), the nine-spine stickleback was the only fish species found in the port lagoon. The port lagoons are closed to the ocean, preventing regular immigration or emigration of fish, have no stream connections, and have been reported to freeze solid in the winter. Because of these factors, coastal lagoons are unlikely to support resident fish communities. Sticklebacks or other fish may wash into the lagoons during storm events. Therefore, pathways to fish in coastal lagoons appear to be incomplete, and lagoon fish are not evaluated in the assessment below.

6.4.1 Coastal Lagoon Benthic Invertebrates Assessment

Potential toxicity to benthic invertebrates of sediments collected in the coastal lagoons to the north and west (prevailing downwind) of the port facilities was evaluated using the 10-day amphipod test based on the amphipod *Hyalella azteca*. Sediment was also evaluated from two coastal lagoons that were south and east of the port facilities (reference lagoons). Percent survival was very high for all of the coastal lagoon sediments, with values of 91 percent or greater. All of the test sediments collected from the coastal lagoons had greater survival than the negative control sediment (i.e., clean sediment provided by the testing laboratory to ensure that the test was performed correctly). Mean survival of amphipods in the negative control sediment was 90 percent. The percentage survival for each sample location is provided in Appendix G (Table G-38). These results indicate that that there are unlikely to be adverse effects to benthic invertebrates in lagoons.

The laboratory report for toxicity testing is provided in Appendix E. A quality assurance review of laboratory procedures and results was conducted by Exponent to ensure that the toxicity tests were consistent with the specifications of the test protocols and that the data are acceptable for use. The complete quality assurance report for the data is provided in Appendix F. Results of the toxicity tests are tabulated in Appendix G, Table G-38. The toxicity testing laboratory report is also included in Appendix G.

6.4.2 Coastal Lagoon Plant Community Assessment

In the following sections, risks to coastal lagoon plant communities are assessed through the interpretation of vegetation community survey data and through comparisons of CoPC concentrations in lagoon sedge samples with literature values representing phytotoxicity thresholds for vascular plants. The focus of the coastal lagoon plant community study was fringing wetland vegetation dominated by graminoids and mare's tail; risks to surrounding tussock tundra may be inferred through the results of the terrestrial plant community assessment. Other vegetation types that may occur near the coast were not evaluated directly in this assessment. A brief description of physical and chemical conditions in coastal lagoons is also provided below.

6.4.2.1 Physical and Chemical Parameters of Coastal Lagoons

Most CoPC concentrations in tundra soil were significantly higher at site lagoons than at the reference lagoons, likely as a result of fugitive dust inputs from the DMTS port (Table 6-3). Station PLNL at the Port Lagoon North had the highest soil concentrations of most CoPCs. Further north at station NLK at the North Lagoon, chemicals such as lead and zinc were still elevated in comparison with reference levels (Table 6-16). Also, the soil pH at PLNL was 1–2 units higher than the reference range (Table 6-16). In the terrestrial environment, soil pH decreased significantly with distance from the road (Table 6-4), and the relatively high soil pH at PLNL, located directly (predominantly) downwind of the DMTS road and port facilities, and the lower pH at more distant lagoon station NLK, is consistent with this trend. In contrast, the lagoon water pH was somewhat lower at PLNL than at NLK or the reference lagoons (7.2 as compared to 8.0–8.6; Table 6-28). Conductivity was up to 13-fold higher at PLNL and up to 4-fold higher at NLK than at the reference lagoons. Conductivity differences may reflect inputs of fugitive dust to the port lagoons. Salinity was also highest at station PLNL (Table 6-28).

6.4.2.2 Plant Community Surveys

Plant communities fringing two site lagoons (Port Lagoon North and the North Lagoon, station locations shown in Figure 4-4) and two reference lagoon communities were surveyed during the 2004 supplemental sampling program, as described above in Section 6.2, *Terrestrial Assessment*. Figure 6-3 diagrams a typical coastal lagoon fringe emergent plant community, showing representative plants observed along the moisture gradient from the lagoon edge to the mesic tussock tundra surrounding the lagoon. Vegetation community parameters and tundra soil characteristics for the four lagoons are summarized in Tables 6-14, 6-15, and 6-16. The lagoon vegetation survey results were presented previously in detail along with the terrestrial plant community survey data in Section 6.2. Section 6.4.2.4, *Risk Characterization for Coastal Lagoon Plants*, incorporates the results of the plant community surveys and the phytotoxicity threshold comparisons presented in the following subsection (Section 6.4.2.3).

6.4.2.3 Comparisons with Phytotoxicity Thresholds

Whole sedge samples (*C. aquatilis* and *E. angustifolium*) were collected from the inner shorelines of site and reference lagoons during the 2004 supplemental sampling program.

Samples were primarily above-ground tissue with some root material, which was rinsed in site water to remove soil or sediment particles. Sedge tops were not washed prior to analysis. Table 6-29 summarizes the CoPC concentrations in whole sedge samples collected at coastal lagoon site and reference stations, and compares the tissue concentrations with phytotoxicity thresholds for vascular plants derived from literature reports, as described previously in Sections 6.2.2 and 6.3.3.2. All CoPC concentrations in lagoon sedge and grass samples were below minimum phytotoxicity thresholds at all site and reference stations (Table 6-29).

6.4.2.4 Risk Characterization for Coastal Lagoon Plants

The plant community survey results did not provide clear evidence of adverse effects to the graminoid community surrounding coastal lagoons as a result of exposure to site-related CoPCs. Differences in plant functional group covers and community indices between site and reference lagoons were not statistically significant (Table 6-3). The dominant vascular plants at stations PLNL and NLK were similar to those at their comparable reference stations, CL-REF-1 and CL-REF-2, respectively (Table 6-14). The relative dominance of species was fairly similar between NLK and CL-REF-2 but differed between PLNL and CL-REF-1. Mare’s tail cover was higher at PLNL, while pendent grass, tundra grass, and cottongrass covers were higher at CL-REF-1 (Table 6-14). These differences may be functions of the position of the vegetation survey line relative to the lagoon shoreline. Figure 6-3 illustrates a typical lagoon vegetation gradient from the waterline upslope to higher tundra areas, where the plant community shifts in response to changes in moisture and other environmental conditions. Because the slope at PLNL was steeper than at CL-REF-1, the survey line at Port Lagoon North was placed very near to the water’s edge (Photograph 19), while the survey line at CL-REF-1 was located about 100–300 ft from the shoreline, in order to try to match the vegetation profiles at the two lagoons (Photograph 20). The proximity of the survey line to open water at PLNL may explain why microplots at that station had high cover values for standing water and mare’s tail, an aquatic plant that grew in monotypic stands near the open water at PLNL and CL-REF-1. It appears that the high mare’s tail (forb) cover at PLNL contributed to this station’s relatively low evenness score (Table 6-15) and isolated this station in the PCA and NMDS plots (Figures 6-5 and 6-6). In contrast, stations NLK and CL-REF-2 had similar diversity and evenness, and they clustered together fairly closely in the PCA plots (Table 6-15; Figures 6-5 and 6-6). The most notable distinction between these two stations was the higher dry blade cover at NLK, but in general, their plant communities were quite similar (Table 6-14).

Comparisons of sedge tissue CoPC concentrations with thresholds for phytotoxicity suggested that the measured CoPC concentrations were not high enough to result in phytotoxicity to dominant coastal lagoon fringe emergent plants such as *Carex* sedges and cottongrass (*E. angustifolium*; Table 6-29). Moss and lichen samples were not collected in the vicinity of the coastal lagoons, and therefore analogous comparisons could not be made for these groups. Moss cover was low at PLNL relative to NLK or the reference lagoon stations (Table 6-14). It cannot be determined from this analysis whether the moss cover results were related to elevated CoPC concentrations at PLNL or were functions of different environmental conditions (e.g., moisture or salinity) at the site and reference stations.

Overall, fringing coastal lagoon vegetation does not appear to be adversely affected by exposures to CoPCs at the site. Plant tissue concentrations did not exceed phytotoxicity thresholds, and qualitative assessments of vegetation vitality did not reveal any evident signs of phytotoxicity. Differences in plant community structure between site and reference communities likely reflect differences in microhabitats where sampling transects were located. Natural variability among and within lagoon plant communities, which fluctuate seasonally in size and composition as water levels rise and recede, may be greater than any site-related differences. Other plant communities occur in the vicinity of the coastal lagoons, some of which may be more sensitive to metals deposition than wetland graminoid communities. These different communities were not surveyed and are not directly evaluated in the risk assessment, and extrapolation of the results for fringing wetland vegetation to other coastal lagoon communities may not be appropriate.

6.5 Wildlife Assessment

Food-web exposure models were developed to estimate site-specific daily doses of CoPCs for avian and mammalian receptors. This approach allows for a direct comparison of exposure rates with measures of toxicity. 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 are used to 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 near the port, 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 fractional intake, 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, a technique used to determine the proportion of the receptor population projected to incur an adverse impact as a result of site-related chemical exposure. Although a probabilistic approach would more accurately characterize the likelihood of various levels of risk to receptor populations than deterministic risk estimates, particularly at a large and diverse site such as this, DEC risk assessment guidance does not allow for use of probabilistic modeling in ERAs (DEC 2000). Therefore, deterministic exposure models were developed for all wildlife receptors, as described in Section 6.5.1, *Exposure Characterization*, and probabilistic exposure modeling was not pursued in the baseline risk assessment. Wildlife exposure estimates were compared with TRVs derived from toxicological studies reported in the scientific literature; TRV derivations are summarized in Section 6.5.2, *Effects Characterization*, and hazard quotient results are reported in Section 6.5.3, *Toxicity Assessment*. The ecological risk characterization for wildlife is presented in Section 6.5.4, *Risk Characterization for Wildlife*.

6.5.1 Exposure Characterization

Using the generic equation described in Section 3.5.6, receptor-specific food-web models were developed to estimate daily dietary exposures to CoPCs for birds and mammals that may feed at the site. Both site and reference scenarios were evaluated for each receptor. These models were used to calculate total dietary exposures to CoPCs resulting from consumption of food and water and the incidental ingestion of sediment or soil on a mg/kg body-weight-day basis. Appendix K provides food-web model worksheets showing concentration inputs and exposure calculations used to assess various exposure scenarios for wildlife. Exposures were calculated for the list of chemicals selected for that receptor in the CoPC screening for wildlife (Sections 3.5.6 and 3.6.3). Based on the results of the ecological screening, 14 chemicals (aluminum, antimony, arsenic, barium, cadmium, chromium, cobalt, lead, mercury, molybdenum, selenium, thallium, vanadium, and zinc) were retained for quantitative evaluation in the baseline ERA for terrestrial and aquatic herbivores, terrestrial invertivores, and terrestrial carnivores. Based on the results of the ecological screening, cadmium, lead, mercury, and zinc were retained as CoPCs for stream invertivores, and cadmium, lead, and zinc were also retained as CoPCs for coastal lagoon invertivores.

Dietary exposures for the baseline assessment are based on a more comprehensive data set than the screening assessment described in Section 3.5.6, in terms of the types of data selected, the prey items sampled, and the spatial extent over which samples were collected. The screening assessment was much more conservative than the baseline assessment. For example, the maximum chemical concentrations reported in food items or environmental media were used in the exposure estimates. The conservative assumptions of the screening represented a worst-case scenario, resulting in protective exposure estimates that were appropriate for a screening-level assessment. The baseline assessment was refined and expanded based on the results of the ecological screening and site-specific observations from the Phase I and Phase II sampling events.

While both newer and older survey data were incorporated into the screening, data from these older surveys were excluded from the baseline analysis in favor of data from more recent sampling events, which better represent current conditions at the site, and which provide more complete data for the list of CoPCs. In the food-web models developed for the baseline ERA, life history information from arctic Alaska was used to select and derive exposure parameters, such as mean body weights and diet compositions, while in the screening ERA, minimum female body weights were used. Water ingestion was not included in the screening assessment exposure analysis, but it was included in the baseline assessment.

In the screening, maximum chemical concentrations in tundra soils were used as a measure of potential exposure via incidental soil ingestion, although the maximum soil and moss concentrations were not necessarily collocated for any chemical. Also, for aquatic habitats, the maximum chemical concentrations from any of the three creeks were used to calculate exposure for fish-eating wildlife. In the baseline exposure assessment, the mean CoPC concentrations were used in all terrestrial and aquatic station-based food-web models. For large home-range receptors foraging within assessment units or across the whole site, mean and 95%UCL on the mean concentrations were calculated.

In the screening, receptors were assumed to be present on site all year, while in the baseline assessment, intake was calculated on a time-use basis, representing the fraction of the year that a receptor may be resident at the site. Both the screening and baseline exposure assessments assumed 100 percent gastrointestinal absorption efficiencies. The exposure parameters and food-web models for the baseline assessment are described in detail below.

6.5.1.1 Exposure Parameters

For all receptors, the food-web models used conservative but ecologically relevant values for exposure parameters (shown in Table 6-30). Whenever available, life history information from arctic Alaska was used to select or derive exposure parameters, such as mean body weights and diet compositions. Food ingestion rates were estimated from measured food consumption rates or energy budgets reported in the literature or were calculated using allometric equations from Nagy et al. (1999). For most receptors, incidental soil or sediment ingestion rates were based on the percentage of soil in wildlife diets reported in Beyer et al. (1994; Table 6-30). Water ingestion rates were derived using drinking water ingestion equations for birds and mammals from U.S. EPA (1993). In the absence of data on relative gastrointestinal absorption efficiencies, the parameter A_i was conservatively set to a value of 1.0 in all models.

Fractional intake (F_i) was calculated on a time-use rather than area-use basis, because many receptors have small home ranges relative to the size of the site (Table 6-30) and would be expected to derive all of their daily diet from the site, or a sub-section of the site, while receptors such as migratory birds are present at the site only for portions of the year and derive the rest of their diet from staging areas and wintering grounds. Thus, a time-use factor representing the fraction of the year that a receptor may be resident at the site was used as the fractional intake parameter. For migratory birds, time use was determined based on first and last sightings of species reported for the Cape Thompson area, which is located approximately 60 miles to the north of the Red Dog port (Williamson et al. 1966; Table 6-30). The residence period of caribou at the site is quite variable, depending on the year and individual, as discussed in Section 6.1.6.1. As a result, estimating an average time-use parameter for this receptor was more ambiguous than for migratory birds. For the purposes of the risk assessment, the caribou was conservatively assumed to occur at the site for 5 months on average based on the possible residence time for an over-wintering individual (Table 6-30). As noted in Section 6.1.6.1, it is likely that much less than one percent of the WACH would over-winter near the site. The majority of caribou are migratory and probably transit the site in a few days to at most a few weeks.

6.5.1.1.1 Small Home-Range Receptors

Daily dietary exposures were modeled on a scale appropriate to the life history of each receptor. Exposures for terrestrial receptors with small home ranges, including the tundra vole, Lapland longspur, and tundra shrew (Table 6-30), were evaluated on a station-by-station basis. Thus, for these receptors, point estimates of dietary exposure were calculated at each site and reference station where appropriate food items were sampled in 2004 (see Section 4).

6.5.1.1.2 Large Home-Range Receptors

Daily dietary exposures for wide-ranging terrestrial receptors that are likely to forage across multiple stations and transects, including the ptarmigan, caribou, moose, snowy owl, and arctic fox (Table 6-30), were modeled for three broad assessment units and the terrestrial reference area (shown in Figure 6-12). The port assessment unit encompassed the area inside the port ambient air boundary and up to 2 km on either side of the DMTS road in the vicinity of the port; the mine assessment unit was defined as the area outside the mine's solid waste boundary and north of Anxiety Ridge Creek; and the road assessment unit comprised the DMTS road corridor between the port and mine assessment units, up to 2 km on either side of the road (Figure 6-12). Caribou and moose exposures were also modeled for the whole site (total of all three assessment units). The assessment unit approach was adopted for these receptors in order to integrate exposure over discrete areas in a manner that reasonably reflects their habitat use but allows for the differentiation of ecological risks along the DMTS road corridor.

6.5.1.1.3 Aquatic Receptors

Exposure estimates for aquatic wildlife were developed for individual streams, tundra ponds, and lagoons sampled in 2004 (Figures 4-1, 4-2, and 4-3). This approach was predicated on the assumption that each water body likely supported discrete individuals of most receptor species, given the spatial segregation of streams in relation to the potential foraging range of individual receptors. For example, muskrat or common snipe foraging at Aufeis Creek would not also be expected to forage at Anxiety Ridge Creek. In this way, risks are estimated for receptors at each stream, pond, and coastal lagoon. Point estimates of exposure were calculated for the green-winged teal, muskrat, moose, and common snipe feeding in streams that cross the DMTS road and in tributaries of Evaingiknuk Creek in the terrestrial reference area (Figure 4-3). Teal and muskrat exposures were also modeled for individual tundra ponds at the site and reference area. Invertebrate tissue data were not collected at tundra ponds during the supplemental sampling in 2004. Therefore, in addition to creek exposure scenarios, the snipe was also evaluated as an invertivore feeding at terrestrial stations rather than tundra ponds, since snipe often forage in terrestrial areas with wet or moist soil. Food-web models for brant and black-bellied plover were developed for two lagoons on site, Port Lagoon North and the North Lagoon, and for two reference lagoons, which are referred to in this text as the Control Lagoon and Reference Lagoon (Figure 4-4).

6.5.1.2 CoPC Concentrations

Measured CoPC concentrations in biota, sediment or tundra soil, and surface water were used to calculate dietary exposures. Table 3-3 summarizes the media data used in the CoPC screening (Section 3.5). With the exception of the marine data and a few older surveys (ENSR91, ENSR92, ENSR95, ENSR96, TECK01, and USGS02), the tundra soil, sediment, and water data used in the CoPC screening were also incorporated into relevant food-web models (Table 3-3). Data from the older surveys were excluded from the analysis in favor of data from more recent sampling events, which better represent current conditions at the site, and which provide more complete data for the list of CoPCs. Data described in Table 3-3 are presented in Appendix C. In addition, tundra soil, sediment, and tissue data collected during the supplemental sampling program in 2004 were used in the exposure models, including metals concentrations in sedge blades, willow and birch leaves, lichens, soil invertebrates, and small mammals collected in the terrestrial environment, and in whole sedge plants, sedge seeds, willow leaves (from shrubs on stream banks), and aquatic invertebrates collected in aquatic environments (described in

Section 4). Analytical results of the 2004 supplemental sampling program are presented in Appendix G. Limited plant data from other investigations were available to model exposures to CoPCs in food for terrestrial herbivores, and these data were also included in food-web models where appropriate. Moss data collected at terrestrial transect stations in 2001 (Exponent 2002a) and 2003 were used in food-web models for receptors that consume a small amount of moss in their diets, including the tundra vole, caribou, and brant (Table 6-30). Moss data collected in and around the CAKR in 2000 and 2001 by NPS (Ford and Hasselbach 2001; Hasselbach 2003, pers. comm.; Hasselbach et al. 2005) and lichen and willow data collected at the site in 2001 (Exponent 2002a) were also used in caribou food-web models. These plant tissue concentrations are presented in Appendices C and G. Only data collected at locations within 2 km of the DMTS road were used in the models to reflect conditions where fugitive dust deposition is greatest. For all data sets, detected and undetected results were used in exposure calculations; undetected values were reported at one-half the detection limit.

Plant and prey tissue data were used to approximate the composition of each receptor's diet, as summarized in Table 6-30. Sedge blade data were used to represent herbaceous plants, and willow and birch leaf data were used to represent shrubs in the diets of terrestrial herbivores such as the willow ptarmigan and tundra vole. In addition, willow leaf samples collected along stream banks and whole sedge samples collected in streams, ponds, and lagoons were included in exposure scenarios for caribou and moose. Soil invertebrate data were used in food-web models for Lapland longspur and tundra shrew. The common snipe was originally selected as a representative receptor for stream and tundra pond avian invertivores. However, the snipe was evaluated as a receptor that foraged in the terrestrial environment in lieu of tundra ponds, because the snipe probes for invertebrates in the tundra soil or temporarily exposed sediments around the edges of ponds and wetlands rather than in submerged sediments (see Section 6.1.2.3). Small mammal data were used to model exposure for snowy owl and arctic fox foraging in the port and road assessment units. Although mammal trapping was attempted on terrestrial transect TT6 near the mine's solid waste boundary (Figure 4-1), no small mammals were caught. Therefore, exposure scenarios could not be developed for owl and fox foraging in the mine assessment unit.

In aquatic systems, whole sedge data were used to model exposures for muskrat, brant, and moose (at coastal lagoons), and sedge seed data were used to model exposures for green-winged teal. No sedge plants were found in Aufeis Creek during the supplemental sampling event, and thus exposure scenarios were not developed for teal and muskrat in this stream. Willow leaf data collected along stream banks were used in exposure models for the moose as an aquatic receptor. Aquatic invertebrate data from streams and coastal lagoons were used in food-web models for common snipe and black-bellied plover, respectively. Invertebrates also constituted 15 percent of the teal's diet, and where available (i.e., streams), aquatic invertebrate data were used in food-web models for teal. Soil invertebrate data were used to model teal exposure in tundra ponds and to fill gaps in the stream data as needed.

In addition to CoPC exposure through food ingestion, incidental soil or sediment ingestion and water ingestion were included in the models. All plant samples were unwashed prior to analysis, and measured CoPC concentrations reflect both the internal tissue concentrations and concentrations in adhered soil or sediment particles. Therefore, including incidental soil or sediment ingestion as separate pathways in the exposure models may represent an overly conservative estimate of exposure to these media for herbivorous wildlife. Tundra soil metals concentrations were used in terrestrial food-web models, and surface sediment concentrations were used in aquatic models. Terrestrial receptors with small home ranges were assumed to drink water from the closest tundra pond, or from the closest stream, in the absence of pond water data. For example, exposure scenarios for receptors feeding at stations along terrestrial transect TT5 near the port (Figure 4-1) were modeled using water data from tundra pond stations TP1-0100 or TP1-1000 (Figure 4-3), while scenarios for receptors feeding along transect TT6 near the mine were modeled using water data from Anxiety Ridge Creek (Figure 4-3). Caribou, arctic fox, and snowy owl were assumed to drink water from ponds or stream stations located within each assessment unit. Aquatic receptors, including lagoon receptors, were assumed to drink from the water body for which the food-web model scenario was developed. Drinking water inputs to all food-web models were total metals concentrations measured in unfiltered samples.

Mean CoPC concentrations were used in all terrestrial station-based food-web models (i.e., models for terrestrial receptors with small home ranges) and all aquatic exposure models. Often, concentration inputs into the station, pond, or creek models were single data points. In cases where multiple results were available, data were averaged across field replicates, sampling events (e.g., tundra soil data from 2003 and 2004) and species within a tissue category (e.g., willow and birch leaves) to derive concentration inputs. Thus, it was assumed that a receptor that browsed on shrubs to obtain part of its diet would eat willow and birch leaves if both were available at a station. However, aquatic herbivores were assumed to eat either whole sedge plants or seeds, depending on feeding styles and food preferences (Sections 6.1.6.2 and 6.1.6.3). For coastal lagoon receptors, station means were calculated first and then averaged to derive mean concentrations for the whole lagoon. Because data sets for terrestrial stations and transects and surface water bodies were small, the 95%UCL on the mean concentrations was not calculated at this spatial scale (i.e., for small home-range receptors).

For large home-range receptors foraging within assessment units or across the whole site, both mean and 95%UCL on the mean CoPC concentrations were calculated, and these statistics were used to derive two sets of exposure estimates for these receptors (mean and 95%UCL on the mean scenarios are provided in Appendix K). Field replicates were averaged, and data were averaged across sampling events. All tissue samples (except field replicates) were treated as individual data points. Each data set was tested for whether it fit a normal, gamma, or lognormal distribution. If these distributions fit, then the appropriate 95%UCL for the fitted distribution was used. If none of these distributions fit, then a non-parametric UCL was used. All UCL calculations were conducted using EPA's ProUCL 3.0 Software, in accordance with EPA exposure point guidance (U.S. EPA 2002d). If the 95%UCL on the mean was greater than the maximum value, the maximum was used instead. Sampling stations located within each assessment unit are summarized in Figure 6-12; Figures 6-13 through 6-19 show station locations by sample type (e.g., tundra soil) and assessment unit.

Complete data sets were not available to model all CoPC exposure scenarios for all receptors. When chemical data were not available for the receptor's primary dietary component, such as small mammals for owl or fox foraging in the mine assessment unit, the scenario was not modeled but is discussed in the uncertainty section (Section 6.6.5.1). In other cases, data were substituted with results from the closest appropriate station. Data sources for food-web model inputs are defined for each exposure scenario in Appendix K.

6.5.2 Effects Characterization

In the CoPC screening for wildlife (Section 3.5.6), worst-case exposure estimates for representative receptors were compared to avian or mammalian NOAEL TRVs to determine the list of CoPCs to be carried forward for evaluation in the baseline risk assessment. Because the NOAEL represents a body-weight-normalized daily intake rate of a chemical that did not elicit any adverse responses in the test organism, exceedance of this value does not necessarily imply that adverse effects would occur for ecological receptors. However, if maximum daily dietary exposures were lower than the NOAEL TRV, then the chemical was not considered likely to cause adverse effects to upper trophic-level receptors and was not retained as a CoPC. In the risk assessment, the estimated daily dietary exposure to a CoPC (described above in Section 6.5.1) is compared against the NOAEL and LOAEL TRV. The LOAEL is the minimum dose reported to elicit a statistically significant adverse effect in the species tested in the pertinent laboratory study. Thus, an exposure rate in excess of the LOAEL TRV may result in an adverse effect to an exposed individual or population. The NOAEL and LOAEL TRVs are used to describe the potential for adverse ecological effects to occur as a result of CoPC exposure.

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), laboratory studies reviewed for TRV derivation were evaluated for the measurement endpoints that are relevant for receptors on a population level: development, reproduction, and survival. Chronic dietary exposure studies were preferred, because they best represent wildlife exposure conditions to CoPCs. For some chemicals with little or no published toxicological information, studies measuring alternate endpoints or with shorter exposure durations had to be used for TRV derivation, as discussed below.

Table 6-31 summarizes the avian and mammalian TRVs that were used in hazard quotient calculations. Based on discussions with DEC and their consultants, two sets of TRVs were used to evaluate avian exposures to zinc (described below), and because arsenic speciation in environmental media at the site is not known, both arsenate- and arsenite-based TRVs were used to evaluate avian and mammalian exposures to arsenic (Exponent 2004b). The form of chromium present at the site has not been analyzed, and therefore mammalian TRVs for hexavalent chromium were used as conservative measures of effects; the uncertainty surrounding this assumption is discussed in Section 6.6.3. Avian chromium TRVs were based on exposure to trivalent chromium, as no suitable TRVs for hexavalent chromium were found. Methylmercury TRVs were selected as effects measures for mercury, because this CoPC is typically in a methylated form in biological tissues (food items), which tend to contribute more mercury to the total exposure than drinking water or incidental ingestion of soil or sediment. No appropriate toxicological studies were found from which to derive avian TRVs for antimony and cobalt, avian LOAEL TRVs for aluminum and vanadium, or a mammalian LOAEL TRV for antimony (Table 6-31). No appropriate studies were identified from which to derive avian or mammalian TRVs for iron or silver. CoPCs without appropriate TRVs cannot be evaluated quantitatively in food-web exposure models, but are discussed later in Section 6.6, *Uncertainty Assessment*. Allometric scaling of TRVs to body weight is not performed in this ERA; however, the implications for risk estimates if allometric scaling were applied are discussed in detail in Section 6.6, *Uncertainty Assessment*. Brief descriptions of TRV derivations are presented below.

6.5.2.1 Aluminum

Carriere et al. (1986) dosed ringed doves with aluminum sulfate in food for 4 months. Because there were no significant reproduction differences observed at a dose of 1,000 ppm over the critical life stage (reproduction), this dose was considered to be an avian no-effect dose. The 1,000-ppm dose was based on wet weight in food and equates to 1,111 ppm dry weight, if a 10 percent moisture content for prepared laboratory food is assumed. Based on a ringed dove food ingestion rate of 0.017 kg/d (calculated with an allometric equation from Nagy 1987) and a body weight of 0.155 kg (Terres 1980), a NOAEL TRV was calculated to be 120 mg/kg-day. No appropriate study could be found to identify an avian LOAEL TRV.

The mammalian TRVs for aluminum were based on a study by Ondreicka et al. (1966). Mice (*Dobra voda* strain), fed a normal laboratory diet containing 170 ppm aluminum, were given drinking water spiked with aluminum chloride. Concentrations were adjusted such that the total intake (19.3 mg/kg-day) was twice that of the dietary intake and exposure continued for 390 days (3 generations). No significant effect was noted with regard to number of litters or number of offspring. However, the treatment group did manifest reductions in weight gain in the second and third litters of the second generation and the first and second litter of the third generation mice. Therefore, the 19.3 mg/kg-day was considered a LOAEL TRV for mammals. Because no lower dose level was tested, a 0.1 level of uncertainty was applied to estimate the no-effects TRV at 1.93 mg/kg-day.

6.5.2.2 Antimony

The no-effects TRV for the evaluation of antimony toxicity to mammals was based on a study by Schroeder et al. (1970). Rats were exposed to 5 ppm antimony potassium tartrate in drinking water over their entire life span. No significant effects were noted with regard to growth, longevity, or reproduction. Using an average rat body mass of 0.35 kg (U.S. EPA 1995b) and intake rate of 0.046 L/day (Calder and Braun 1983), a NOAEL TRV of 0.66 mg/kg-day was determined. No appropriate study could be found to identify a mammalian LOAEL TRV or avian TRVs.

6.5.2.3 Arsenate

In a study by Stanley et al. (1994), arsenic (as sodium arsenate) was fed to mallards (*Anas platyrhynchos*) in the diet for 115−128 days during reproduction at arsenic dose levels of 0, 25, 100, or 400 mg/kg. Arsenic did not affect hatching success or embryo deformity rates at any dose level; however, the highest dose resulted in an increase in the number of days between pairing and laying of the first egg, and a decrease in whole egg weight and shell thickness. Duckling production and growth decreased when diets were supplemented with 400 mg/kg arsenic. Thus, 400 mg/kg arsenic in the diet represented a LOAEL dose, whereas 100 mg/kg arsenic represented a NOAEL dose. Assuming a mallard body weight of 1.0 kg (Heinz et al. 1989) and an ingestion rate of 0.100 kg/day (Heinz et al. 1989), the arsenate NOAEL TRV was calculated to be 10 mg/kg-day, and the LOAEL to be 40 mg/kg-day.

Nemec et al. (1998) evaluated the developmental toxicity of arsenic (arsenate in arsenic acid H_3AsO_4, 52.8 percent arsenic by weight) to rabbits (New Zealand white strain). Rabbits were provided arsenic acid by oral gavage on gestation days 6 through 18 at 0, 0.19, 0.75, or 3.0 mg/kg-day and sacrificed on gestation day 29. Maternal effects including mortality (7 of the 20 does), slight decreases in body weight, and clinical signs of toxicity occurred only at the highest dose level. There were no statistically significant effects on embryos or fetuses at this dose, although there was a slight decrease in the number of viable fetuses per litter. No maternal or offspring effects were seen at 0.75 mg/kg-day, which is equivalent to a NOAEL dose of 0.40 mg arsenic/kg-day when adjusted for the proportion of arsenic in the arsenic acid compound. The 3 mg/kg-day dose represented a chronic LOAEL, which, when adjusted, equates to 1.6 mg arsenic/kg-day.

6.5.2.4 Arsenite

In a USFWS (1964) study, mallards were exposed to 100, 250, 500, and 1,000 ppm sodium arsenite (57.67 percent As^{+3}) in their diet for 154 days (128 days for the 100-ppm group). Ducks in the 100-ppm group experienced no mortality. Ducks in the 250-ppm group experienced 12 percent mortality, but ducks in the control groups experienced an average of 13 percent mortality. Ducks in the 500-ppm group experienced 60 percent mortality. The daily consumption rate for birds in the 250-ppm dose group was 34 mg sodium arsenite (57.67 percent arsenic by weight) per kg body weight, or 20 mg arsenic per kg body weight. Because the average mortality in this group was no greater than the average mortality in the corresponding controls, 20 mg/kg-day was considered a chronic NOAEL. The 500-ppm dose was considered a LOAEL dose because of the elevated mortality. The daily intake rate in this group was 86 mg sodium arsenite per kg body weight, which equates to a LOAEL of 50 mg/kg-day arsenic.

Mammalian TRVs for arsenic were developed from a study by Schroeder and Mitchener (1971). Mice were exposed to 5 ppm arsenite in drinking water over three generations. The treatment had no significant effect on either progeny mortality or fertility, but did produce a slight but significant suppression in productivity. A LOAEL TRV of 1.3 mg/kg-day was derived based on a drinking water intake rate of 0.0075 L/d for a 30 g mouse (Calder and Braun 1983). The LOAEL TRV was multiplied by a LOAEL-to-NOAEL uncertainty factor of 0.1 to derive a NOAEL TRV of 0.13 mg/kg-day.

6.5.2.5 Barium

The LOAEL and NOAEL TRVs for barium toxicity to birds were developed from a study by Johnson et al. (1960). Day-old chicks were exposed to eight dose levels (250, 500, 1,000, 2,000, 4,000, 8,000, 16,000, and 32,000 ppm) of barium hydroxide in food for 4 weeks. Some mortality occurred at the 4,000-ppm dose. This dose was considered the LOAEL dose. The LOAEL TRV was calculated by assuming a mean body weight of 0.121 kg, a food consumption rate of 0.0126 kg/day (U.S. EPA 1988), and a subchronic to chronic uncertainty factor of 0.1. The LOAEL is 42 mg/kg-day. The highest treatment resulting in no clinical signs was 2,000 ppm. Therefore, a NOAEL TRV of 21 mg/kg-day was calculated with the same assumptions stated above.

The NOAEL TRV calculation for barium in mammals was based on an investigation performed by Perry et al. (1983). In this study, rats were exposed to barium chloride at three dose levels (1, 10, and 100 ppm in water) in drinking water for 16 months. The test species had a body weight of 0.435 kg and a water consumption rate of 0.022 L/day. None of the three dose levels affected food or water consumption or growth, but rats exposed to 10 or 100 ppm barium exhibited cardiovascular hypertension. The significance of hypertension to potential adverse effects in wildlife populations is unclear. Therefore, the maximum no effects dose of 100 ppm that did not affect other functions (i.e., food or water consumption or growth) was used to calculate a chronic no-effects TRV of 5.1 mg/kg-day for the evaluation of risk to mammals.

The LOAEL TRV distribution for barium toxicity to mammals was developed from a study by Borzelleca et al. (1988). Rats were exposed to 300 mg/kg body weight-day barium chloride (198 mg/kg-day barium) through oral gavage in water for 10 days. This treatment resulted in increased relative kidney masses and reduced ovarian masses. Because of the short duration of this study, the LOAEL TRV was developed by applying a 0.1 uncertainty factor to derive a value of 20 mg/kg-day.

6.5.2.6 Cadmium

White and Finley (1978) exposed mallards to cadmium chloride at three dose levels (1.6, 15.2, and 210 ppm in food) for 90 days through reproduction. The test species had a body weight of 1.153 kg and a food consumption rate of 0.110 kg/d. Mallards exposed to the two lower dosages exhibited no adverse effects, but those exposed to 210 ppm cadmium in food produced significantly fewer eggs than did the other groups. Therefore, 15.2 ppm cadmium (1.45 mg/kg-day) was considered to be a chronic NOAEL TRV and 210 ppm cadmium (20 mg/kg-day) was considered to be a LOAEL TRV for birds.

Sutou et al. (1980) exposed rats to cadmium, as cadmium chloride, at four dose levels (0, 0.1, 1.0, and 10 mg/kg-day) by oral gavage through mating and gestation (6 week exposure period). Adverse reproductive effects (i.e., reduced fetal implantations, reduced fetal survivorship, and increased fetal resorptions) were observed in the rats exposed to 10 mg/kg-day. Therefore, the 1 mg/kg-day dose was considered to be a chronic no-effects TRV and a dose of 10 mg/kg-day was considered the LOAEL TRV for the evaluation of risk to mammals.

6.5.2.7 Chromium

The TRV for chromium exposure in birds was based on the study by Haseltine et al. (1985, as cited in Sample et al. 1996). Black ducks were exposed to chromium(III) (as $CrK(SO_4)_2$) at two dose levels (10 and 50 ppm chromium[III]) in food for 10 months (through reproduction). No effects on reproduction were observed at the lower dose of 10 ppm chromium[III] (11 ppm dry weight). The assumptions used in the TRV calculations included a body weight of 1.25 kg (Dunning 1993) and a food consumption rate of 0.0785 kg/kg-day for the test species (based on a reasonable maximum energy [RME] requirement of 200 kcal/kg-day derived from Nagy [1987], an assimilation efficiency of 80 percent, and an energy content of 3,190 kcal/kg dry weight). Therefore, the NOAEL TRV was determined to be 0.86 mg/kg-day. The LOAEL was determined to be 4.32 mg/kg-day based on the 50 ppm treatment.

The NOAEL TRV for mammalian receptors was developed based on a study by MacKenzie et al. (1958). In this study, rats were exposed to chromium[VI] (as $K_2Cr_2O_4$) at six dose levels in drinking water (0.45, 2.2, 4.5, 7.7, 11.2, and 25 ppm chromium[VI]) for 1 year. At the end of this period of exposure, changes in liver, kidney, and bone mass were compared between the treatments. No adverse effects were observed at any of the dose levels. Therefore, the maximum dose (25 ppm chromium in water) was considered to be a chronic NOAEL TRV for mammals. Assuming a rat body weight of 0.35 kg and a water consumption rate of 0.046 L/day (U.S. EPA 1988), this dose corresponds to a TRV of 3.3 mg/kg-day.

The LOAEL for exposure to chromium for mammals was based on a study by Gross and Heller (1946). Rats were exposed daily to 1,250, 2,500, 5,000, or 10,000 mg/kg potassium chromate (as chromium(VI)) in their diets for 3 months. Subnormal offspring (not defined in the study) were reported for rats treated with 2,500 mg/kg chromium. Based on a food concentration of 669.5 ppm (the proportion of chromium (VI) in potassium chromate), an average body mass for the test species of 168 g (as reported in the study), and an intake rate of 0.0172 kg/day (U.S. EPA 1988), and applying a subchronic-to-chronic uncertainty factor of 0.1, a LOAEL of 69 mg/kg-day was determined.

6.5.2.8 Cobalt

Mammalian TRVs for cobalt are based on a study by Nation et al. (1983). Rats were exposed to cobalt in the diet at rates of 5 and 20 mg/kg-day for 69 days. Rats exposed to 20 mg/kg-day performed slower in neurological tests (both reward and aversion tests) than the control animals. This was also accompanied by significant testicular atrophy in males. The animals exposed at a rate of 5 mg/kg-day showed no significant difference in the neurological test battery, nor did they manifest the testicular atrophy. Because of the short duration of the study, and uncertainty factor of 0.1 was used in the derivation of the TRVs. The NOAEL TRV for cobalt exposure was therefore determined to be 0.50 mg/kg-day, and the LOAEL was determined to be 2.0 mg/kg-day.

6.5.2.9 Lead

Pattee (1984) dosed American kestrels with metallic lead in the diet (0, 10, or 50 ppm) for 5−7 months prior to and during clutch completion. Key results of this study included no effects on body weight, food consumption, clutch initiation, interval between eggs, clutch size, fertility, or eggshell thickness at any dose level. Results indicated that the highest tested dose (50 ppm) represented a no effects level. Because the dosing lasted 7 months and included a critical lifestage (reproduction), the study can be considered a chronic exposure. Using a food ingestion rate of 0.01 kg/day and a body weight of 0.13 kg (Sample et al. 1996), a NOAEL TRV of 3.9 mg/kg-day was calculated.

In a study by Edens et al. (1976), Japanese quail received dietary exposure to lead (0, 1, 10, 100, or 1,000 ppm as lead acetate) from hatching to 12 weeks of age, through reproduction. The key result of this study was the observation of a significant decrease in percent hatch of settable eggs at 100 ppm and higher (59.1 percent for this dose group versus 81.6 percent for control group and 82.4 percent for the 10-ppm group). Therefore, 100 ppm lead was considered to be a chronic LOAEL dose. Assuming a body weight of 0.15 kg from Vos et al. (1971) and a food consumption rate of 0.0169 kg/day (based on allometric equation from Nagy 1987), a LOAEL TRV of 11 mg/kg-day was derived.

Mammalian TRVs for lead were developed from a study by Azar et al. (1973) that examined effects on reproductive performance in rats over three generations. Various dose levels were tested (5, 18, 62, 141, 1,130, and 2,102 ppm lead (as lead acetate) measured in food). None of the lead dose levels affected the number of pregnancies, number of live births, or other reproductive indices. The two highest doses reduced offspring weights and produced kidney damage in young. Therefore, 1,130 mg/kg in food, or 90 mg/kg-day (based on a body weight of 0.35 kg and an ingestion rate of 0.028 kg/day from U.S. EPA 1988), was considered the LOAEL TRV. The no-effects dose was 141 mg/kg in food, which corresponds to a TRV of 11 mg/kg-day.

6.5.2.10 Mercury

The TRV used to evaluate the effects of methylmercury in birds was based on a 3-generation study by Heinz (1974, 1976a,b, 1979) in mallards. Mallard ducks were exposed to dietary concentrations of methylmercury dicyandiamide ranging from 0.5 to 3.0 mg/kg dry weight for two generations, with the third generation exposed to 0.5 mg/kg-day. The initial test birds (P1) showed no behavioral or reproductive effects at the lowest methylmercury concentration. However, the second-generation ducklings (F2), demonstrated a 29 percent reduction in 1-week survival rates at 0.5 mg/kg methylmercury (Heinz 1976a). Neither the first generation (F1) nor the third generation (F3) showed decreased survival at this dose level. The impact over the three generations was reported to be an 18-percent reduction in productivity overall. Based on a food intake rate of 128 g/kg body weight (as reported by Heinz 1979), and a body weight of 1.0 kg for the treated F1 and F2 females, this represents a LOAEL of 0.064 mg/kg body weight-day. No long-term studies were identified as suitable for the derivation of a no-effects level for methylmercury exposure to birds. Therefore, an uncertainty factor of 0.5 was applied to estimate a NOAEL TRV of 0.032 mg/kg-day from the LOAEL as recommended by U.S. EPA (1995b).

Sample et al. (1996) applied an LOAEL-to-NOAEL uncertainty factor of 0.1 to derive the NOAEL of 0.0064. U.S. EPA (1995a), when deriving TRVs from the same study, used an uncertainty factor of 0.5 “because the LOAEL appeared to be very near the threshold for effects of mercury on mallards.” According to U.S. EPA (1995a) the recommended range for uncertainty factor is 0.1 to 1. Furthermore, U.S. EPA (1995a) states that, “In cases where a NOAEL cannot be quantified and only an unbounded LOAEL is available, determination of the appropriate value for the uncertainty factor must be done on a chemical specific and test-specific basis with the use of best professional judgment.” U.S. EPA (1995a) provides additional guidance that a larger value for the uncertainty factor (closer to 1) could be used for an unbounded LOAEL that is judged to be at or near the dose-response threshold for the endpoint being evaluated. Data from the Heinz (1979) study showed that the reduction in productivity occurred in only one of the three generations, indicating that the LOAEL dose is near the dose-response threshold. Therefore, based on the limited–magnitude-of-effects-seen study that is the basis of this TRV, a 10-fold uncertainty factor as used by Sample et al. (1996) is an overly conservative estimate of the true no-effects threshold, and the 2-fold factor as recommended by U.S. EPA (1995b) is more appropriate.

The toxicity of methylmercury to mammals was based on a study by Verschuuren et al. (1976). Rats were dosed with three dose levels of 0.1, 0.5, and 2.5 ppm of methylmercury chloride in food. The study took place over three generations and reproduction was used as the toxicity endpoint. Adverse effects were not observed at the two lower doses, although exposure to 2.5 ppm reduced pup viability. The 0.5-ppm dose was considered the no-effect dose, and with a body weight of 0.35 kg (U.S. EPA 1988) and a food consumption rate of 0.028 kg/day (U.S. EPA 1988), the NOAEL was calculated to be 0.032 mg/kg-day. The lowest-effect dose was considered to be 2.5 ppm, and the LOAEL was calculated to be 0.16 mg/kg-day.

6.5.2.11 Molybdenum

A study by Lepore and Miller (1965) was used to determine the NOAEL and LOAEL for birds. In this study, chickens were dosed with three levels of molybdenum (500, 1,000, and 2,000 ppm, as sodium molybdate) in their food for 21 days during reproduction. Embryonic viability was reduced at the 500-ppm level of treatment. This dose was considered to be the LOAEL dose, and using a body weight of 1.5 kg (U.S. EPA 1988) and a food consumption rate of 0.106 kg/day, the LOAEL was calculated to be 35 mg/kg-day. The NOAEL (3.5 mg/kg-day) was estimated by multiplying the LOAEL with a LOAEL-to-NOAEL uncertainty factor of 0.1.

Schroeder and Mitchener (1971) exposed mice to 10 mg/L and 0.45 mg/kg of molybdenum (as MoO_4) in water and food, respectively. The experiment lasted three generations, for more than 1 year. Reproductive success with high incidence of runts was observed at this dose. The LOAEL (2.6 mg/kg-day) was calculated with the 0.45 mg/kg dose in food, 10 mg/L dose in water, a food ingestion rate of 5.5 g/day, a water ingestion rate of 0.0075 L/day, and a body weight of 0.03 kg. The NOAEL (0.26 mg/kg-day) was estimated by applying a LOAEL-to-NOAEL uncertainty factor of 0.1.

6.5.2.12 Selenium

The toxicity of selenium to birds was evaluated based on the results of a study by Heinz et al. (1989). Mallard ducks were fed selenium (as selenomethionine) from 0 to 16 mg/kg-day in the diet for 100 days prior to egg set. Reproductive productivity was significantly reduced in the 8 ppm (8.8 ppm dry weight) treatment with no significant effects reported at 4 ppm (4.4 ppm dry weight). Based on an average body weight of 1 kg and a food intake rate of 90 g dry weight/day (Heinz et al. 1987), a NOAEL TRV of 0.4 mg/kg-day and a LOAEL TRV of 0.8 mg/kg-day were derived.

The evaluation of selenium toxicity to mammalian receptors is based on a study by Rosenfeld and Beath (1954) where rats were exposed to three levels of potassium selenate (1.5, 2,5, or 7.5 ppm) in drinking water over two generations. The treatment group exposed to 2.5 ppm showed no significant difference with regard to reproduction or number of young reared. However, the second-generation female progeny of this treatment group did show a 50 percent reduction in the number of young reared. Therefore, the no-effects TRV was determined based on a dose of 1.5 ppm. Assuming a water intake rate of 0.046 L/day (based on the scaling function of Calder and Braun 1983) and an average body weight of 0.35 kg (U.S. EPA 1988), a NOAEL TRV of 0.20 mg/kg-day and a LOAEL TRV of 0.33 mg/kg-day were determined.

6.5.2.13 Thallium

The evaluation of thallium toxicity in mammals was based on a study by Formigli et al. (1986). In this investigation, rats were exposed to 10 mg/L thallium sulfate in water for 60 days. The study dosage resulted in reduced sperm motility, and was therefore considered a LOAEL. Applying the reported water intake rate of 0.270 mg/rat, and average body mass of 0.365 kg, a corresponding LOAEL of 0.74 mg/kg was determined. The NOAEL TRV was therefore estimated by applying a 0.1 uncertainty factor to this toxicity estimate to derive a value of 0.074 mg/kg-day.

Limited information was found on the effects of thallium in birds. Hudson et al. (1984) reported mortality in mallards (*Anas platyrhynchos*), ring-necked pheasant (*Phasianus colchicus*), and golden eagle (*Aquila chrysaetos*) following acute oral gavage exposure to thallium sulfate. Final mortality counts were made after a 14-day observation period following treatment, except when test animals showed signs of intoxication past 14 days. Hudson et al. (1984) reports LD_{50} values ranging from 23.7 mg/kg (pheasant) to 60.0 mg/kg (golden eagle). The lowest reported LD_{50} of 23.7 mg/kg, reported for ring-necked pheasants, provided the basis for the thallium LOAEL TRV (24 mg/kg-day). An acute–to-chronic uncertainty factor of 0.1 and a LOAEL-to-NOAEL uncertainty factor of 0.1 were applied to yield a NOAEL TRV for birds of 0.24 mg/kg-day.

6.5.2.14 Vanadium

The NOAEL for birds was developed from a study by White and Dieter (1978). In this study, mallard ducks were dosed with 2.84, 10.36, and 110 ppm of vanadium (as vanadyl sulfate) in food for 12 weeks. The researchers observed endpoints such as mortality, body weight, and blood chemistry, and found that no adverse effects were observed at any of the dose levels. Therefore, a NOAEL of 11 mg/kg-day was calculated based on the dose of 110 mg/kg, a food ingestion rate of 121 g/day, and a body weight of 1.17 kg. A LOAEL TRV could not be calculated from this study, and no other studies were identified that could be used to derive a LOAEL.

The mammalian TRV was developed based on a study by Domingo et al. (1986). In this investigation, rats were exposed to sodium metavanadate ($NaVO_3$) at three dose levels (5, 10, and 20 mg/kg-day at 41.78 percent vanadium) by oral intubation. Exposure started 60 days prior to gestation and continued through gestation, delivery, and lactation. Significant adverse effects (i.e., increased number of stillbirths per litter, decreased offspring size and weight) were observed at all dose levels. Therefore, the lowest dose (2.09 mg/kg-day vanadium by percentage of weight) was considered to be the chronic LOAEL. The NOAEL TRV for mammals was therefore determined by applying a 0.1 uncertainty factor to yield a value of 0.209 mg/kg-day.

6.5.2.15 Zinc

The avian TRV for zinc toxicity was based on a dietary feeding study performed by Stahl et al. (1990). In this study, 24- or 56-week-old white leghorn hens were exposed to zinc sulfate in the diet from 28 mg/kg (control) to 2,000 mg/kg in a dehydrated corn and soybean meal diet. After continuous daily exposure to 68 weeks of age, no significant differences were noted in hen weight, feed consumed, egg production, egg fertility, egg hatchability, or progeny growth rates. Sample et al. (1996) states that in the study by Stahl et al. (1990), there was a reduction in egg hatchability at the highest dose (2,000 mg/kg zinc) and used that dose to derive a LOAEL of 130 mg/kg-day. However, hatchability was reduced in only one of two studies conducted by Stahl et al. (1990). In study 1, the hatchability of birds feeding on a diet containing 2,000 mg/kg zinc was higher than the hatchability of control birds (85.9 percent versus 81.5 percent). In study 2, hatchability of the zinc-fed birds was 69.8 percent versus 86.5 percent for the control birds. Stahl et al. (1990) stated "the fertility and hatchability of the eggs incubated during the two studies were not affected significantly by the level of Zn in the diet." Additionally, no significant differences were noted in hen weight, feed consumed, egg production, or progeny growth rates. The authors concluded, "The zinc treatments have no effect on hen performance or reproductive performance." Therefore, given the minor level of effects noted, and the lack of statistically significant differences between control and treatment groups, Sample's classification of the highest dose as representing a LOAEL is unsupported by results of the study. This dose more accurately represents a NOAEL, and therefore, 130 mg/kg-day is the appropriate NOAEL TRV (calculated with a dietary concentration of 2,000 mg/kg, and a measured intake rate of 0.06 kg dry weight/kg body weight, assuming 10 percent moisture content).

A study by Jackson et al. (1986) did find significant adverse effects (reduced food consumption, body weight, and egg production) for a diet with 2,000 mg/kg zinc added to basal concentrations, but no adverse effects for a diet with 1,000 mg/kg zinc added. The chronic NOAEL and LOAEL TRVs derived from this study were 70 and 124 mg/kg-day, respectively. The NOAEL TRV was based on 1,056 mg/kg of zinc in food, a food intake rate of 124.3 g/day and a body weight of 1.87 kg. The LOAEL TRV was based on a zinc concentration of 2,056 mg/kg in food, food intake rate of 107.1 g/day, and body weight of 1.78 kg (all body weights and food ingestion rates as reported in the study). The results of the Stahl et al. (1990) and Jackson et al. (1986) papers indicate that dose levels of 1,000 mg/kg to 2,000 mg/kg appear to bracket the true effects threshold, at least in chickens, and depending on the endpoint chosen, either dose could be the NOAEL. Therefore, both sets of TRVs are used to evaluate risks to birds in the ERA.

The NOAEL and LOAEL TRVs used to evaluate risks from zinc exposure in mammals were developed from a study by Schlicker and Cox (1968). In this investigation, adult female Sprague-Dawley rats were exposed to 2,000 and 4,000 mg/kg dry weight zinc oxide in their diets. Exposure commenced 21 days prior to mating and continued throughout gestation. Females exposed to 4,000 ppm exhibited increases in fetal resorption. No effect on reproduction (measured as percent resorption or difference in rate of fetal growth) was observed at 2,000 ppm. Based on an assumed body mass of 0.35 kg (U.S. EPA 1988) and a food ingestion rate of 0.028 kg/day, the LOAEL and NOAEL TRVs were calculated to be 160 and 320 mg/kg-day, respectively.

6.5.3 Toxicity Assessment

To assess the potential for adverse ecological effects to occur in bird and mammal populations, the exposure and effects characterizations were integrated 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).

For every food-web model exposure scenario at the site, the daily dietary exposure to a CoPC was compared against the NOAEL and LOAEL TRVs (Table 6-31). Estimated daily exposures in reference areas were compared to the same TRVs to assess the ecological risks to receptors from exposure to chemical concentrations that occur naturally in the environment and to provide a context for evaluating incremental risks to receptors that are exposed to elevated metals concentrations at the site. For migratory receptors that spend only a portion of the year at the site, hazard quotients for the site and reference areas were weighted by residence period (time-use, reported in Table 6-30) and summed to derive a quotient that reflected year-round chemical exposure. Although these receptors likely encounter a range of chemical concentrations in their winter ranges, estimated daily exposures in the terrestrial reference area or reference lagoons were used as reasonable approximations for offsite dietary exposures. When more than one reference scenario had been developed for a receptor, the scenario with the most complete data or the highest chemical concentrations (most conservative scenario) was selected, including station ST-REF-5 for evaluation of green-winged teal in streams; station TP-REF-3 for evaluation of teal in tundra ponds; station ST-REF-3 for evaluation of common snipe in streams; and the Reference Lagoon for evaluation of brant and black-bellied plover in coastal lagoons (Figures 4-3 and 4-4). In caribou exposure models, the mean and 95%UCL on the mean reference exposures were used to calculate mean and 95%UCL on the mean year-round hazard quotients, respectively.

The focus of the toxicity assessment is on receptor and chemical combinations for which hazard quotients suggest the potential for adverse ecological effects (i.e., hazard quotients greater than 1.0). The majority of receptor and chemical combinations evaluated in the risk assessment had NOAEL-based hazard quotients below 1.0, indicating a low likelihood of adverse ecological effects. Detailed data and results for all receptor exposure scenarios are presented in

Appendix K. Summaries of hazard quotient modeling results for terrestrial, freshwater aquatic, and coastal lagoon receptors are presented in the following sections.

6.5.3.1 Terrestrial Environment

Food-web models for terrestrial receptors were developed to estimate dietary exposures at individual stations or assessment units, depending on the receptor's home range size, as previously discussed in Section 6.5.1. Thus, hazard quotients for receptors with small home ranges, such as the tundra vole and Lapland longspur, are presented for terrestrial stations (e.g., TT5-0010) where food items were sampled. Plant tissues were collected more broadly than terrestrial invertebrates during the supplemental sampling program, and therefore more risk evaluation scenarios could be developed for terrestrial herbivores (20 site stations and 3 reference stations) than terrestrial invertivores (13 site stations and 1 reference station; refer to Section 4 for a summary of the supplemental data collection). Hazard quotients for ptarmigan, caribou, moose, and terrestrial carnivores, which have large home ranges, are presented by ecological assessment unit (e.g., port, road, and mine; see Figure 6-12) and also for the terrestrial reference area.

6.5.3.1.1 Willow Ptarmigan

The willow ptarmigan represents terrestrial avian herbivore populations in the risk assessment (Table 6-1). As described in Section 6.5.1.2, mean and 95%UCL on the mean exposure scenarios were calculated for ptarmigan and compared to NOAEL and LOAEL TRVs. For eight CoPCs (aluminum, arsenic, cadmium, chromium, molybdenum, selenium, thallium, and vanadium), all hazard quotients in all assessment units were less than 1.0. Hazard quotients that exceeded 1.0 for ptarmigan are summarized in Table 6-32. Mean exposures to barium exceeded the NOAEL TRV in the road and mine assessment units (hazard quotients of 1.2 and 1.9, respectively). Exposures to the 95%UCL on the mean barium concentrations exceeded the NOAEL TRV at the road (hazard quotient of 1.7) and exceeded the NOAEL and LOAEL TRVs at the mine (hazard quotients of 4.0 and 2.0). Mean exposures to lead exceeded the NOAEL TRV at the port and mine (hazard quotients of 2.4 and 1.6, respectively), and 95%UCL on the mean exposures to lead exceeded NOAEL and LOAEL TRVs (hazard quotients of 6.2 and 2.2 at the port and 3.5 and 1.2 at the mine). The 95%UCL on the mean exposure to mercury in the port assessment area slightly exceeded the NOAEL TRV (hazard quotient of 1.2). The 95%UCL on the mean exposures to zinc at the port and mine also exceeded the NOAEL TRV (hazard quotients of 1.3 and 1.4, respectively). However, mean exposures did not exceed the LOAEL TRV for any CoPC. Hazard quotients for willow ptarmigan did not exceed 1.0 in the terrestrial reference area.

6.5.3.1.2 Tundra Vole

The tundra vole represents small-bodied terrestrial mammalian herbivores in the risk assessment (Table 6-1). For eight CoPCs (antimony, cadmium, chromium, cobalt, mercury, selenium, thallium, and zinc) all hazard quotients at all sampling locations were less than 1.0. Figure 6-20 shows the distribution of hazard quotients greater than 1.0 for tundra vole. Hazard quotients tended to decrease with distance from the road and were generally lower in the central portion of the DMTS transportation corridor than in the vicinity of the port or mine. NOAEL-based hazard quotients for aluminum, arsenic (arsenite), barium, lead, molybdenum, and vanadium exceeded 1.0 for tundra vole at one or more site stations (Figure 6-20). Aluminum and barium exposures exceeded their LOAEL TRVs at all 10-m and 100-m stations except for barium at TT2-0100 (Figure 6-20). The magnitude of exceedances was generally low, especially for barium. At 10-m stations, hazard quotients ranged from 3.1–21 for aluminum and from 1.6–5.0 for barium. At 100-m stations, hazard quotients ranged from 1.4–7.9 for aluminum and from 1.1–3.3 for barium. LOAEL-based hazard quotients at 1,000-m and 2,000-m stations were generally less than 1.0, with slight exceedances for aluminum at station TT6-1000 (1.3) and TT7-1000 (1.6), and for barium at TT7-1000 (1.2) and TT7-2000 (1.3). These stations are downwind of the mine's solid waste boundary (Figure 6-20). No other chemicals had LOAEL-based hazard quotients above 1.0 at the site.

In the terrestrial reference area, NOAEL-based hazard quotients for aluminum and barium also exceeded 1.0, and the LOAEL-based hazard quotient for aluminum was 3.0 at station TS-REF-5 (Figure 6-20). Compared with reference results, hazard quotients for aluminum were elevated at 10-m stations but were generally within the reference range at 100-m stations (except TT6-0100), 1,000-m stations, and 2,000-m stations (Figure 6-20). Hazard quotients for barium were elevated over reference results at 10-m and 100-m stations near the port and in the middle of the DMTS road, and at all stations along transects TT6 and TT7 near the mine (Figure 6-20).

6.5.3.1.3 Caribou

The caribou represents large-bodied terrestrial mammalian herbivores in the risk assessment (Table 6-1). For 11 CoPCs (antimony, arsenic, cadmium, chromium, cobalt, mercury, molybdenum, selenium, thallium, vanadium, and zinc) all hazard quotients were less than 1.0. Hazard quotient results exceeding 1.0 for caribou are presented in Table 6-33. Mean and 95%UCL on the mean exposures to aluminum exceeded their NOAEL and LOAEL TRVs in all assessment units, and hazard quotients were fairly uniform across the site. For example, NOAEL-based hazard quotients ranged from 22–25 and LOAEL-based hazard quotients ranged from 2.2–2.5, both based on mean exposure scenarios (Table 6-33). Mean aluminum exposure in the terrestrial reference area also exceeded its NOAEL TRV (hazard quotient of 8.9), and at the 95%UCL on the mean, aluminum exposure exceeded both its NOAEL and LOAEL TRVs in the reference area, with hazard quotients of 16 and 1.6, respectively (Table 6-33). However, aluminum hazard quotients were about 2- to 3-fold as high as at the site as in the reference area (Table 6-33). Mean and 95%UCL on the mean exposures to barium exceeded their NOAEL TRV in all assessment units, with hazard quotients ranging from 1.6–5.1 for the mean exposure scenario and from 2.6–7.9 for the 95%UCL on the mean exposure scenario. For the mean exposure scenario, barium slightly exceeded the LOAEL TRV in the mine assessment unit (hazard quotient of 1.3), where barium exposures were highest (Table 6-33). The 95%UCL on the mean exposure to barium slightly exceeded its LOAEL TRV at the mine (hazard quotient of 2.0) and across the whole site (hazard quotient of 1.2, Table 6-33). Lead exposure at the 95%UCL on the mean slightly exceeded its NOAEL TRV in the mine assessment unit (hazard quotient of 1.1) but did not exceed its LOAEL TRV in this area (Table 6-33). Mean lead exposures did not result in any hazard quotients greater than 1.0 for caribou. No other chemical exposures for caribou exceeded TRVs at the site or reference area.

6.5.3.1.4 Moose

The moose also represents terrestrial mammalian herbivore populations in the risk assessment (Table 6-1). While caribou was modeled as a migratory receptor, moose was evaluated as a permanent resident at the site. Of the 14 CoPCs evaluated, only aluminum exceeded its TRVs. Mean and 95%UCL on the mean aluminum exposures exceeded their NOAEL TRV in all assessment units at the site. Mean exposures to aluminum resulted in NOAEL-based hazard quotients for moose of 1.4 at the port, 1.3 along the road, 1.2 at the mine, and 1.5 across the whole site. Aluminum exposures at the 95%UCL on the mean for moose resulted in hazard quotients of 2.3 at the port, 3.0 along the road, 1.7 at the mine, and 2.9 across the whole site. Mean and 95%UCL exposures to aluminum in the reference area also exceeded their NOAEL TRV, and the reference hazard quotients (1.1 and 2.5, respectively) were comparable to site values. No chemical exposures for moose exceeded their LOAEL TRV.

6.5.3.1.5 Lapland Longspur

The Lapland longspur represents terrestrial avian invertivore populations in the risk assessment (Table 6-1). No hazard quotients exceeded 1.0 for any of the 12 CoPCs evaluated for Lapland longspur at terrestrial stations at the site or reference area.

6.5.3.1.6 Common Snipe

The snipe represents terrestrial avian invertivores in the risk assessment (Table 6-1). For 9 CoPCs (arsenic, cadmium, chromium, mercury, molybdenum, selenium, thallium, vanadium, and zinc), all hazard quotients were less than 1.0. The NOAEL-based hazard quotients for aluminum were slightly greater than 1.0 at stations located approximately 10 m from the DMTS road, including TT5-0010 (1.2), TT2-0010 (1.2), and TT3-0010 (1.1), and at station TS-REF-5 (1.3) in the terrestrial reference area. NOAEL-based hazard quotients for lead were also slightly greater than 1.0 at stations TT5-0010 (1.5) and TT5-0100 (1.3) near the port. Barium exposures slightly exceeded their NOAEL TRV at stations TT6-0010 (1.7) and TT6-0100 (1.7) near the mine's solid waste boundary. No LOAEL-based hazard quotients for common snipe exceeded 1.0 at the site or in the terrestrial reference area.

6.5.3.1.7 Tundra Shrew

The tundra shrew represents terrestrial mammalian invertivore populations in the risk assessment (Table 6-1). Four CoPCs (antimony, chromium, cobalt, and thallium) had hazard quotients less than 1.0 at all locations. Figure 6-21 shows the distribution of hazard quotients greater than 1.0 for tundra shrew. In general, the number of chemicals with hazard quotients above 1.0 and the magnitude of the exceedances tended to decrease with distance from the road. NOAEL-based hazard quotients for aluminum, arsenic (arsenite), barium, cadmium, lead, mercury, selenium, vanadium, and zinc exceeded 1.0 at one or more site stations (Figure 6-21). Of these chemicals, aluminum and barium also had some LOAEL-based hazard quotients above 1.0 at the site, with hazard quotients of 0.54–8.8 for aluminum and 0.11–7.2 for barium. At station TT5-0010, the LOAEL-based hazard quotient for selenium was 1.0 (Figure 6-21). Exposures to aluminum and barium exceeded their LOAEL TRVs at all 10-m and 100-m stations but were below their LOAEL TRVs at 1,000-m and 2,000-m stations, with the exception of barium at TT6-1000 (Figure 6-21).

In the terrestrial reference area, NOAEL-based hazard quotients for aluminum, barium, selenium, and vanadium exceeded 1.0, and the LOAEL-based hazard quotient for aluminum also exceeded 1.0 (Figure 6-21). Exposure to aluminum was higher in the reference area than at any site station, whereas exposure to barium was higher at 10-m and 100-m stations than in the reference area (Figure 6-21). At site stations where selenium and vanadium exceeded their NOAEL TRVs, hazard quotients were less than 2-fold higher than comparable reference results (Figure 6-21).

6.5.3.1.8 Snowy Owl

The snowy owl represents terrestrial avian carnivore populations in the risk assessment (Table 6-1). Snowy owl exposures were modeled for port and road assessment units and the terrestrial reference area using mean and 95%UCL on the mean CoPC concentrations (no small mammal data were available for the mine assessment unit). Of the 12 CoPCs evaluated, only mercury had hazard quotients greater than 1.0. Hazard quotients for mercury did not exceed 1.0 in the port assessment unit or the reference area. However, mean exposures to mercury in the road assessment unit resulted in a NOAEL-based hazard quotient of 3.3 and a LOAEL-based hazard quotient of 1.7 (95%UCL on the mean results were 14 and 7.2, respectively).

6.5.3.1.9 Arctic Fox

The arctic fox represents terrestrial mammalian carnivore populations in the risk assessment (Table 6-1). Like the snowy owl models, fox exposure scenarios were developed for the port and road assessment units and the terrestrial reference area using mean and 95%UCL on the mean CoPC concentrations. Of the 14 CoPCs evaluated for arctic fox, only aluminum and mercury had hazard quotients greater than 1.0, as summarized in Table 6-34. The other 12 CoPCs (antimony, arsenic, barium, cadmium, chromium, cobalt, lead, molybdenum, selenium, thallium, vanadium, and zinc) all had hazard quotients less than 1.0. Mean and 95%UCL on the mean exposures to aluminum exceeded their NOAEL TRV in the road assessment unit and both their NOAEL and LOAEL TRVs in the port assessment unit (Table 6-34). For mean exposure scenarios, NOAEL-based hazard quotients were 2.8 and 11 at the road and port, respectively. The corresponding LOAEL-based hazard quotients were 0.28 and 1.1. Mean exposure to mercury exceeded its NOAEL TRV in the road assessment unit (hazard quotient of 2.6), and the 95%UCL on the mean exposure to mercury exceeded both its NOAEL and LOAEL TRVs in the road assessment unit (Table 6-34). In the reference area, NOAEL-based hazard quotients for aluminum exceeded 1.0 but were below site hazard quotients (Table 6-34).

6.5.3.2 Freshwater Aquatic Environments

Food-web models for freshwater aquatic receptors were developed for three stream stations located near the DMTS road and four individual tundra ponds at the site. In addition, reference exposure scenarios were developed for three creek stations and three tundra ponds located in the terrestrial reference area (Figure 4-3). Risks to herbivores were assessed in streams (Omikviorok River and Anxiety Ridge Creek) and ponds, and risks to invertivores were assessed in streams only, as invertebrate data were not available in ponds (see Section 6.5.3.1.6 for snipe evaluation in the terrestrial environment).

6.5.3.2.1 Green-winged Teal

The green-winged teal represents avian herbivores feeding in streams and tundra ponds in the DMTS road corridor (Table 6-1). No hazard quotients for the teal exceeded 1.0 in streams or ponds at the site. Aluminum and chromium exposures at reference pond station TP-REF-5 slightly exceeded the NOAEL TRV, with hazard quotients of 1.2 and 1.1, respectively. No hazard quotients for green-winged teal exceeded 1.0 in reference streams.

6.5.3.2.2 Muskrat

The muskrat represents small-bodied mammalian herbivores that might forage in streams and tundra ponds at the site (Table 6-1). Of the 14 CoPCs evaluated, 9 (antimony, cadmium, chromium, lead, mercury, molybdenum, selenium, thallium, and zinc) had no hazard quotients greater than 1.0 at the site. Figure 6-22 shows the distribution of hazard quotients greater than 1.0 for muskrat. In the Omikviorok River, located in the central portion of the DMTS road, NOAEL-based hazard quotients for aluminum, arsenic (arsenite), barium, and vanadium exceeded 1.0, but only aluminum exposure exceeded its LOAEL TRV (hazard quotient of 8.3; Figure 6-22). In Anxiety Ridge Creek, located near the mine's solid waste boundary, aluminum and barium exposures exceeded their NOAEL TRVs, aluminum exposure also exceeded its LOAEL TRV (hazard quotient of 1.8), and barium exposure was equal to its LOAEL TRV (Figure 6-22). Aluminum, arsenic (arsenite), and barium exposures also exceeded their NOAEL TRVs in one or more reference streams, and aluminum exposures exceeded their LOAEL TRV in all three reference streams with hazard quotients of 1.3–2.5 (Figure 6-22). Of the chemicals with hazard quotients greater than 1.0 in streams, arsenic exposure in the Omikviorok River and aluminum exposure in Anxiety Ridge Creek were within the ranges estimated for reference stations, while other hazard quotients were about two to four times as high at the site as in the reference area (Figure 6-22).

In tundra ponds, aluminum, barium, and cobalt exposures exceeded their NOAEL TRVs at one or more site stations (Figure 6-22). Barium exposure at station TP4, located near the mine's solid waste boundary, slightly exceeded its LOAEL TRV (hazard quotient of 1.1; Figure 6-22). In the reference area, NOAEL-based hazard quotients for aluminum, arsenic (arsenite), barium, chromium, and vanadium exceeded 1.0 at one or more stations (Figure 6-22). When compared with reference results, cobalt exposure at TP1-1000 and barium exposure at TP4 appeared to be elevated, but other TRV exceedances in site ponds were within the ranges estimated for reference ponds (Figure 6-22).

6.5.3.2.3 Moose

The moose was evaluated as a terrestrial receptor (Section 6.5.3.1) and as a representative receptor for large-bodied mammalian herbivores feeding along stream channels (Table 6-1).

Only aluminum and barium had NOAEL-based hazard quotients greater than 1.0 for moose in streams, and no LOAEL-based hazard quotients exceeded 1.0 for this receptor. Hazard quotient results for aluminum were 3.1 in Aufeis Creek, 5.0 in Omikviorok Creek, and 3.0 in Anxiety Ridge Creek. Aluminum exposures in reference streams also exceeded their NOAEL TRV, resulting in hazard quotients of 1.1 at stream reference station ST-REF-3 and 2.9 at stations ST-REF-5 and ST-REF-6 (food-web models for both reference stations used sediment data from ST-REF-5, resulting in similar hazard quotients for aluminum; see Appendix K). Barium exposure in Anxiety Ridge Creek slightly exceeded its NOAEL TRV (hazard quotient of 1.2), but all other hazard quotients for barium were less than 1.0.

6.5.3.2.4 Common Snipe

The common snipe represents avian invertivores feeding in streams and on the tundra (see Section 6.5.3.1.6; Table 6-1). No CoPCs had hazard quotients greater than 1.0 for snipe in site or reference streams.

6.5.3.3 Coastal Lagoons

Food-web models for coastal lagoon receptors were developed for two site lagoons and two reference lagoons (Figure 4-4). Risks to avian and mammalian herbivores and avian invertivores were assessed in this environment.

6.5.3.3.1 Brant

The brant represents coastal lagoon avian herbivore populations in the risk assessment (Table 6-1). Of the 12 CoPCs evaluated, no hazard quotients for brant exceeded 1.0 at site or reference lagoons.

6.5.3.3.2 Muskrat

The muskrat represents small-bodied mammalian herbivores that may feed on coastal lagoon vegetation. Hazard quotients for all chemicals except aluminum were less than 1.0 for the muskrat. NOAEL-based hazard quotients for aluminum were 4.8 in Port Lagoon North, 7.6 in the North Lagoon, 9.3 in the Reference Lagoon, and 9.7 in the Control Lagoon. Exposures did not exceed the LOAEL TRV for aluminum.

6.5.3.3.3 Moose

The moose represents large-bodied mammalian herbivores that may forage in and around the coastal lagoons. Aluminum exposures exceeded the NOAEL TRV, but not the LOAEL TRV, in all site and reference lagoons, but hazard quotients for all other CoPCs were less than 1.0. Aluminum hazard quotients were higher in the Reference and Control Lagoons (2.3 and 2.4, respectively) than in Port Lagoon North or the North Lagoon (1.2 and 1.9, respectively).

6.5.3.3.4 Black-bellied Plover

The black-bellied plover represents coastal lagoon avian invertivore populations in the risk assessment (Table 6-1). Exposure estimates for plover resulted in one lead TRV exceedance at the site. The NOAEL-based hazard quotient for lead was 1.4 at Port Lagoon North, but the LOAEL-based hazard quotient for lead (0.48) did not exceed 1.0. No other CoPCs had hazard quotients greater than 1.0 at Port Lagoon North, and all hazard quotients for black-bellied plover at the North Lagoon, Control Lagoon, and the Reference Lagoon were below 1.0.

6.5.3.4 Hazard Quotient Summary

In summary, estimated daily dietary exposures to CoPCs for some avian and all mammalian receptors exceeded NOAEL or LOAEL TRVs in at least one site location. However, food-web exposure model results for Lapland longspur, green-winged teal (as a stream and pond receptor), common snipe (as a stream receptor), and brant did not exceed any TRVs. Exposure estimates for moose (as a terrestrial, stream, and coastal lagoon receptor), common snipe (as a terrestrial receptor), muskrat (as a coastal lagoon receptor), and black-bellied plover did not exceed any LOAEL TRVs. Hazard quotients for three CoPCs (antimony, chromium, and thallium) did not exceed 1.0 for any wildlife receptors foraging at the site.

Avian exposures to barium, lead, and mercury at the site resulted in some NOAEL-based hazard quotients that were greater than 1.0 and higher than comparable reference hazard quotients. Ptarmigan exposures to barium and lead and snowy owl exposure to mercury also exceeded their LOAEL TRVs. In the absence of appropriate TRVs, avian exposures to antimony and cobalt could not be evaluated using the hazard quotient approach, but are addressed later in the Uncertainty Assessment.

For mammalian receptors as a group, exposures to 11 CoPCs (aluminum, arsenic, barium, cadmium, cobalt, lead, mercury, molybdenum, selenium, vanadium, and zinc) at the site resulted in NOAEL-based hazard quotients that were greater than 1.0 and higher than reference area hazard quotients. Small mammal (tundra vole and tundra shrew), caribou, and muskrat exposures to aluminum and barium exceeded their LOAEL TRVs in at least one site location, and fox exposures to aluminum and mercury at the site also exceeded their LOAEL TRVs.

6.5.4 Risk Characterization for Wildlife

In this section, hazard quotient results for each receptor are evaluated and interpreted to characterize the ecological risks to assessment endpoints (survival, growth, and reproduction of wildlife populations), particularly the incremental risks incurred by populations exposed to CoPCs at the site over those of reference populations. Two sets of risk calculations were performed to derive hazard quotients in Section 6.5.3, *Toxicity Assessment*: daily dietary exposure estimates for representative receptors were compared against 1) no-effects levels (NOAEL TRVs), and 2) thresholds at which significant adverse effects to test organisms were observed in laboratory studies (LOAEL TRVs). Exposure estimates that are below the NOAEL TRV identify conditions under which adverse ecological effects are unlikely to occur to bird or mammal populations, where populations are described as the animals within each assessment unit, because members of those populations are exposed to CoPC levels known through observation to cause no significant effects in test organisms.

Exposure estimates greater than the NOAEL TRV, but less than the LOAEL TRV indicate that individuals are ingesting chemicals in excess of a toxicity threshold and may exhibit adverse effects similar to those observed in the test organisms. In these cases, risk cannot definitively be concluded to be negligible, because the true effect threshold is not exactly known, only that it lies somewhere between the NOAEL and LOAEL. Furthermore, because the endpoints measure organism-level responses, there is considerable uncertainty regarding how these effects, if occurring, would translate to population-level demographics.

For CoPCs where hazard quotients are greater than 1.0 in comparison to both the NOAEL and LOAEL TRVs, adverse effects could occur in wildlife receptors, and could affect population-level parameters (e.g. survivorship, productivity, population abundance, etc). However, if a hazard quotient is less than or equal to hazard quotients for the same receptor-CoPC exposure scenario in the reference area, then it can be concluded that the site poses no incremental risk over background exposures in that case.

Both the dietary exposure estimates calculated in Section 6.5.1 and the toxicity thresholds derived in Section 6.5.2 have inherent uncertainties associated with their assumptions, and as point estimates of exposure and toxicity, they do not fully capture the natural variability in receptor populations and the environment. In addition, as noted above, the extrapolation from predicted toxicity to exposed individuals to population-level effects is imprecise. The uncertainty assessment presented in Section 6.6 evaluates the uncertainties introduced into the analysis by different components of the risk assessment, including uncertainties with predictions of population-level effects. In Section 6.7, *Interpretation of Ecological Significance*, risk estimates for wildlife are discussed together with plant and invertebrate community assessments to evaluate their overall significance to the ecosystems surrounding the DMTS corridor, in light of the inevitable uncertainties associated with the risk analyses.

6.5.4.1 Terrestrial wildlife

The following subsections provide the risk characterization discussion for terrestrial wildlife receptors, including willow ptarmigan, tundra vole, caribou, moose, Lapland longspur, tundra shrew, snowy owl, and arctic fox.

6.5.4.1.1 Willow Ptarmigan

All hazard quotients for aluminum, arsenic, cadmium, chromium, molybdenum, selenium, thallium, and vanadium were below 1.0 for ptarmigan. Exposures to these chemicals would be very unlikely to result in adverse effects to herbivorous birds. Exposures to 95%UCL on the mean concentrations of mercury at the port and zinc at the port and mine exceeded the NOAEL TRVs. However, hazard quotients for mercury and zinc were fairly low (1.2–1.4), and mean exposures did not exceed NOAEL TRVs (Table 6-32). Based on the food-web model results, dietary exposure to mercury or zinc is unlikely to result in adverse effects to herbivorous birds, but risk cannot definitively be concluded to be negligible for the most exposed individuals in the population.

Exposures to barium at the port were below the NOAEL TRV and therefore unlikely to adversely affect individuals residing near the coast. Exposures to barium in the road assessment unit exceeded the NOAEL TRV, with hazard quotients up to 1.7 (based on the 95%UCL exposure), but did not exceed the LOAEL TRV (Table 6-32). However, in the mine assessment unit, the mean and 95%UCL on the mean exposures to barium approached or exceeded the LOAEL TRV (hazard quotients of 0.94 and 2.0, respectively), indicating that herbivorous birds foraging near the mine may experience adverse effects from barium exposure.

LOAEL-based hazard quotients for lead were 0.84 and 2.2 in the port assessment unit and 0.55 and 1.2 in the mine assessment unit (based on mean and 95%UCL on the mean concentrations, respectively). These results suggest that adverse effects from lead exposure are possible near the port and mine.

6.5.4.1.2 Tundra Vole

Hazard quotient results for the tundra vole (Figure 6-20) indicate that voles inhabiting tundra near the road or near the mine's solid waste boundary are at higher risk from exposure to some CoPCs (particularly aluminum, arsenic, barium, lead, and vanadium) than voles inhabiting the reference area. The results showed incremental risk from aluminum, barium, and vanadium exposure at 10 m from the road. Lead exposure near the port and mine and arsenic exposure near the mine also exceeded reference exposures (Figure 6-20). Most chemical exposures decreased to no-effects levels or were comparable to reference exposures by 100 m from the road and 1,000 m from the mine's solid waste boundary. Barium exposure was at or near reference levels by 1,000 m at the port and in the middle of the DMTS road corridor but remained elevated at all stations near the mine (Figure 6-20). Lead exposure dropped below the NOAEL TRV by 1,000 m at the port and 2,000 m at transect TT7 (Figure 6-20).

Incremental risk from chemical exposure near the DMTS road and mine does not necessarily translate into unacceptable ecological risk to vole populations over a broader spatial scale, however. For example, arsenic (as arsenite) and vanadium exposures were greater than NOAEL TRVs and reference area exposures at some 10-m stations, but not substantially greater. The hazard quotients were relatively low (all less than 2.0), arsenic exposures did not exceed their arsenate NOAEL TRV, and arsenic and vanadium exposures did not exceed their LOAEL TRVs (Figure 6-20). Based on these results, there is a possibility that individual voles could exhibit adverse effects from arsenic or vanadium exposures near the road or mine. While these effects, if present, could affect key attributes of voles near the road (e.g., survival, reproduction), it is uncertain what effect, if any, this could have on vole populations, particularly beyond 10–100 m from the road, which would be dependent on factors such as the magnitude of effects near the road, and whether that area acts as a population sink that attracts voles from more distant areas where exposure is lower. Likewise, lead exposures exceeded no-effects levels but not LOAEL TRVs at four stations near the port and mine (Figure 6-20), and hazard quotients were fairly low (maximum of 2.6) and decreased with distance from sources. The results indicate that if adverse effects occur to voles from exposure to these CoPCs, they are most likely to exist in localized areas near facilities, but may not affect the tundra vole population existing at areas beyond about 100 to 1,000 m from the port facilities or mine boundary.

Aluminum and barium TRV exceedances were more widespread than for other CoPCs. However, aluminum exposures were within the range of reference exposures at stations farther than 10 m from the road or mine's solid waste boundary (except at TT6-0100) and therefore do not indicate incremental ecological risks at those distances. Although aluminum hazard quotients at 10 m were elevated, the TRVs used to evaluate exposure may be conservatively low, as described below in the uncertainty assessment for wildlife (Section 6.6.3.4), and thus likely overestimate actual effects. The frequency of elevated hazard quotients for barium across the site also appears to result from the conservative TRVs applied in the risk calculations; these TRVs likely overestimate risks to mammal populations, as discussed in the uncertainty section (Section 6.6.5.4).

6.5.4.1.3 Caribou

Food-web exposure models indicated that caribou foraging in all assessment units were exposed to higher aluminum and barium concentrations in their diets than caribou in the reference area, and that mean exposure levels exceeded the LOAEL TRV for aluminum and the NOAEL TRV for barium in all areas (Table 6-33). The risk calculations for the most exposed individuals in the caribou population (those that consumed food, soil, and water with 95%UCL on the mean CoPC concentrations) predicted LOAEL-based hazard quotients of 4.0 to 6.7 for aluminum and 0.66 to 2.0 for barium in site assessment units (Table 6-33). The 95%UCL on the mean lead exposure in the mine assessment unit slightly exceeded its NOAEL TRV (hazard quotient of 1.1), but all other hazard quotients for lead were less than 1.0 (Table 6-33). Thus, food-web exposure models suggest the possibility for adverse effects to individual caribou from exposure to these three CoPCs, particularly for the individuals most highly-exposed to aluminum or barium.

Exposure scenarios are based on the assumption that caribou over-winter near the DMTS. As discussed in Section 6.1.6.1, surveys conducted by DFG have shown that caribou density is very low during winter in the region of western Alaska that includes the Red Dog mine. Based on 17 consecutive years of data, the maximum density is about 2.6 individuals/square mile, and in many years no individuals are present (DFG 2003c). Even at the highest density, only about 85 caribou would be predicted to over-winter within 1 km of the 52-mile-long DMTS road corridor. The population of the WACH is estimated at about 430,000 individuals; thus, even using the most conservative assumptions regarding over-wintering density, less than 0.02 percent of the population may be subject to the exposure scenarios presented in this risk assessment. Therefore, even if possible adverse effects may occur to a few individuals, there is little likelihood of any corresponding effect at the population level.

Substantially greater numbers of caribou can occur near the mine during migration, although DFG notes that individual caribou spend relatively little time near the Red Dog development complex (DFG 2003c). Craighead and Craighead (1987) found that satellite-tracked migrating caribou moved about 9–12 km (5.6–7.5 miles) per day. At this speed, a migrating caribou could potentially traverse the length of the DMTS road in 7–9 days. If models are run assuming a migrating caribou spends 9 days on the site, then risk associated with exposure is much lower than estimated for an over-wintering individual. For aluminum, the LOAEL-based hazard quotients range from 0.97–0.99 for mean exposure scenarios and from 1.7–1.9 for the 95%UCL on the mean exposure scenario. These values are comparable to reference area hazard quotients (0.89 for mean exposure scenario, 1.6 for 95%UCL on the mean exposure scenario), indicating that very little incremental risk exists to migrating caribou. For barium, LOAEL based hazard quotients for the mean exposure scenario range from 0.16 to 0.21 compared with a reference area hazard quotient of 0.14, again indicating that the incremental risk associated with migration through the mine site is very low. For lead, the NOAEL-based TRVs for migratory caribou are all less than 0.1. Therefore, adverse effects would not be predicted for caribou passing through the mine area during migration.

6.5.4.1.4 Moose

Hazard quotients for moose foraging on the tundra were low. Only aluminum exposures exceeded their NOAEL TRV. The maximum hazard quotient for aluminum was 3.0 (95%UCL scenario for the road), and site hazard quotients were less than or just slightly elevated over reference hazard quotients. Estimated mean and 95%UCL on the mean chemical exposures for moose on the tundra did not reach observed or predicted adverse effects levels and were generally below no-effects levels. The risk calculations suggest that there is a low likelihood of adverse ecological effects to resident large-bodied mammalian herbivore populations exposed to site-related chemicals in the tundra.

6.5.4.1.5 Lapland Longspur

Hazard quotients for the Lapland longspur were all less than 1.0 for all CoPCs, indicating that dietary exposure to CoPCs is very unlikely to result in adverse ecological effects to terrestrial avian invertivores at the site.

6.5.4.1.6 Common Snipe

In the snipe terrestrial assessment, NOAEL-based hazard quotients for aluminum were slightly greater than 1.0 at stations located approximately 10 m from the DMTS road, including TT5-0010 (1.2), TT2-0010 (1.2), and TT3-0010 (1.1), and at station TS-REF-5 (1.3) in the terrestrial reference area. NOAEL-based hazard quotients for lead were also slightly greater than 1.0 at stations TT5-0010 (1.5) and TT5-0100 (1.3) near the port. Barium exposures exceeded their NOAEL TRV at stations TT6-0010 (1.7) and TT6-0100 (1.7) near the mine's solid waste boundary. No LOAEL-based hazard quotients for common snipe exceeded 1.0 at the site or in the terrestrial reference area.

Common snipe exposures to lead in the terrestrial environment exceeded their NOAEL TRV at stations TT5-0010 and TT5-0100 near the port, and snipe exposures to barium exceeded their NOAEL TRV at stations TT6-0010 and TT6-0100 near the mine's solid waste boundary. However, the NOAEL-based hazard quotients were low (less than 2.0) in each case, and no exposures exceeded their LOAEL TRVs. Therefore, the potential for adverse effects to snipe populations from lead or barium exposure is low, although risk cannot definitively be concluded to be negligible. Common snipe exposures to aluminum also exceeded their NOAEL TRV at 10-m stations; however, the aluminum hazard quotient was higher at reference station TS-REF-5 than at any site stations. The hazard quotient results suggest that there is no incremental risk to terrestrial avian invertivores from aluminum exposure at the site. These results are consistent with the findings for Lapland longspur, another avian invertivore, which also showed a low likelihood of ecological risk from CoPC exposure (discussed in the previous section).

6.5.4.1.7 Tundra Shrew

Exposure results for tundra shrew showed the most TRV exceedances of any receptor in the risk assessment, but most of these hazard quotients were relatively low (<3.0) and were based on comparisons to no-effects levels (Figure 6-21). Although exposures to nine chemicals exceeded their NOAEL TRVs at site stations, only aluminum and barium exposures exceeded their LOAEL TRVs, and hazard quotients for aluminum were less than the reference hazard quotient (Figure 6-21). The high reference hazard quotient for aluminum was driven by the soil concentration, as shown in Appendix K. Elevated hazard quotients for barium appear to be partially a function of the TRVs used to evaluate exposures, as discussed in the uncertainty assessment for wildlife (Section 6.6.5.4). The results of the risk calculations for tundra shrew indicate that although risk cannot definitively be concluded to be negligible, exposure to CoPCs at the site is unlikely to result in broad-scale population-level effects to terrestrial mammalian invertivores, although localized effects could occur. Barium, which was elevated across the site and had NOAEL-based hazard quotients greater than 20 near the mine (Figure 6-21) may have adverse effects on shrews in localized areas, such as near the mine.

6.5.4.1.8 Snowy Owl

Mercury was the only CoPC with a hazard quotient exceeding 1.0 for the snowy owl. Food-web exposure models predicted ecological risks to snowy owls exposed to 95%UCL on the mean mercury concentrations and possibly mean concentrations in the road assessment unit (LOAEL-based hazard quotients of 7.2 and 1.7, respectively). High mercury (13.5 mg/kg dry weight, versus 0.03–0.36 mg/kg for the other road samples) in one small mammal sample collected at station TT3-0100 drove these risk estimates. Dixon's Outlier tests, based on log-transformed concentrations, showed that the high mercury result was an outlier with respect to mercury concentrations in all small mammal samples collected onsite ($p<0.01$) and with respect to concentrations in samples from the road assessment unit only ($p<0.05$) (Dixon 1953). When the outlier is excluded from exposure calculations, 95%UCL on the mean and mean hazard quotients for mercury drop to 0.40 and 0.21, respectively, based on the NOAEL TRV, and 0.20 and 0.11, respectively, based on the LOAEL TRV. Mercury hazard quotients were also less than 1.0 in the port assessment unit but were not calculated for the mine assessment unit, because small mammal tissue data were not available for that area. Although the small mammal tissue data show some individuals may have elevated mercury concentrations (Appendix G), overall levels in small mammal populations at the port and along the road are not sufficiently elevated to result in population-level effects to terrestrial avian carnivores.

6.5.4.1.9 Arctic Fox

Mercury and aluminum were the only CoPCs with hazard quotients exceeding 1.0 for the arctic fox. Aluminum exposure estimates for arctic fox exceed their NOAEL TRV in the port and road assessment units and in the reference area, and exposures at the port also exceeded their

LOAEL TRV (Table 6-34). However, the LOAEL-based hazard quotients at the port were only 1.1 for individuals exposed to average CoPC concentrations and 2.1 for the most exposed individuals (Table 6-34). Aluminum hazard quotients for the road assessment unit were slightly higher than reference, but exposures did not exceed their LOAEL TRV (Table 6-34). Given the conservative nature of the aluminum TRVs for mammals (discussed in Section 6.6.5.4), this relatively low level of risk is not likely to translate into population-level ecological effects to terrestrial mammalian carnivores. Food-web model scenarios for fox also predicted incremental risk from mercury exposure in the road assessment unit (Table 6-34). However, when the small mammal outlier was excluded from the risk calculations (see the risk characterization for snowy owl), hazard quotients for 95%UCL on the mean and mean mercury exposures dropped to 0.31 and 0.17, respectively, based on the NOAEL TRV, and 0.061 and 0.033, respectively, based on the LOAEL TRV. Although the fox may encounter occasional prey items with high mercury concentrations along the DMTS road, average mercury exposure for fox was below the LOAEL TRV along the road (hazard quotient of 0.51; Table 6-34) and was low at the port (NOAEL-based hazard quotient of 0.056; Appendix K), suggesting that adverse effects to individuals from mercury exposure are not likely to have a significant impact on the site's terrestrial mammalian carnivore populations.

6.5.4.2 Freshwater Aquatic Wildlife

The following subsections provide the risk characterization discussion for freshwater aquatic wildlife receptors, including green-winged teal, muskrat, moose, and snipe.

6.5.4.2.1 Green-winged Teal

Food-web exposure model results for green-winged teal did not exceed NOAEL TRVs in any site streams or ponds. Therefore, dietary exposure to CoPCs is not predicted to cause adverse ecological effects to freshwater aquatic avian herbivores using the DMTS road corridor.

6.5.4.2.2 Muskrat

Muskrat exposures to aluminum, arsenic, barium, cobalt, and vanadium exceeded NOAEL TRVs at one or more stream or tundra pond stations at the site (Figure 6-22). However, exposures to arsenic (arsenite), cobalt, and vanadium are not likely to cause adverse ecological effects to the muskrat population inhabiting the larger, entire assessment area, because TRV exceedances for these chemicals were isolated occurrences, and the exposure estimates were less than LOAEL TRVs (Figure 6-22). Arsenic and vanadium exposure estimates in reference freshwater bodies also exceeded NOAEL TRVs and in some cases were greater than site exposures (Figure 6-22), indicating that risk levels for these chemicals are no greater at the site than in the reference area. Aluminum hazard quotients were within the range calculated for reference streams and ponds, except at the Omikviorok River station (Figure 6-22). Barium hazard quotients were elevated over reference results in both site streams and in tundra pond TP4, but LOAEL-based hazard quotients were at or near 1.0 in these scenarios (Figure 6-22). Therefore, there is little incremental ecological risk to the site's aquatic mammalian herbivore populations from aluminum exposure but some incremental risk from barium exposure, although the conservative nature of the barium TRVs (see Section 6.6.5.3) suggests that significant effects to populations from barium exposure are unlikely.

6.5.4.2.3 Moose

Moose exposure to aluminum in the Omikviorok River channel resulted in a NOAEL hazard quotient of 5.0 and a LOAEL hazard quotient of 0.50, while hazard quotient results in Aufeis and Anxiety Ridge Creeks (3.1 and 3.0, respectively) were very close to hazard quotients in two out of three reference streams (1.1, 2.9, and 2.9), and therefore represented no incremental risks to moose (Appendix K). Given that the NOAEL-based hazard quotients for aluminum in site streams were equal to or no more than twice the reference hazard quotients, and that no exposure estimates exceeded their LOAEL TRV, it is unlikely that exposure to aluminum would cause significant incremental risk to moose foraging in site streams, especially when uncertainties associated with the aluminum TRV (see Section 6.6.5.4) are considered. Barium exposure in Anxiety Ridge Creek slightly exceeded its NOAEL TRV (hazard quotient of 1.2) but was less than the LOAEL TRV (hazard quotient of 0.30), and no other exposure models predicted barium hazard quotients greater than 1.0 for moose. Thus, dietary exposure to CoPCs in site streams is not expected to cause adverse ecological effects to large-bodied terrestrial herbivore populations using the DMTS road corridor, although risks from aluminum and barium cannot definitively be concluded to be negligible.

6.5.4.2.4 Common Snipe

Hazard quotients for the common snipe were all less than 1.0 in site streams, indicating that dietary exposure to CoPCs is very unlikely to result in adverse ecological effects to freshwater aquatic avian invertivores at the site.

6.5.4.3 Coastal Lagoon Wildlife

The following subsections provide the risk characterization for coastal lagoon wildlife receptors, including brant and black-bellied plover.

6.5.4.3.1 Brant

No hazard quotients for any CoPC in the Port Lagoon North or the North Lagoon exceeded 1.0 for brant. The low risk estimates for brant indicate that exposures to CoPCs in coastal lagoons are unlikely to cause adverse ecological effects to seasonal populations of avian herbivores at the DMTS port.

6.5.4.3.2 Muskrat

Only one CoPC (aluminum) had NOAEL-based hazard quotients greater than 1.0 for muskrat, and hazard quotients were greater in the reference lagoons than in the site lagoons. Therefore, exposure to CoPCs in the site lagoons does not result in incremental risk to herbivorous mammals such as muskrats.

6.5.4.3.3 Moose

Hazard quotient results for moose were similar to the results for muskrat: aluminum exposures exceeded the NOAEL TRV in all lagoons, and hazard quotients were higher in reference lagoons than in site lagoons. Exposures to other CoPCs did not exceed TRVs. Thus, the risk results for moose support the conclusion that exposure to CoPCs is unlikely to cause adverse effects to herbivorous mammals in the coastal lagoon environment.

6.5.4.3.4 Black-bellied Plover

Hazard quotient results for black-bellied plover at coastal lagoons suggest a very low likelihood of adverse ecological effects from exposure to cadmium or zinc, which did not exceed their NOAEL TRVs. Lead exposure in Port Lagoon North slightly exceeded its NOAEL TRV (hazard quotient of 1.4) but was less than the LOAEL TRV (hazard quotient of 0.48), and hazard quotients were elevated over reference lagoon results (0.93 and 0.33 in the Reference Lagoon and 0.20 and 0.07 in the Control Lagoon). Lead exposure in the North Lagoon was less than the NOAEL TRV (hazard quotient of 0.91) and within the reference range. Thus, while plovers or other shorebirds feeding in Port Lagoon North may be exposed to higher lead than shorebirds feeding in lagoons off the site, the predicted exposure is below the known adverse effects threshold and would not likely result in unacceptable ecological risk to aquatic avian invertivore populations at the DMTS port, although risk cannot definitively be concluded to be negligible.

6.6 Uncertainty Assessment

There are a number of inherent uncertainties associated with any risk assessment. Uncertainties can exist with regard to the characterization of CoPC concentrations in site media and biota, or with the interpretation of the ecological significance of those CoPC concentrations on receptor populations. This section presents a detailed evaluation of most important sources of uncertainty and the effects of these uncertainties on conclusions about the extent and magnitude of risks. There are several major sources of uncertainty related to results of the DMTS ERA:

- Uncertainties related to reference area selection
- Uncertainties related to sample size
- Uncertainties related to CoPC screening
- Uncertainties related to the terrestrial assessment
  - Uncertainty in the plant community surveys
  - Uncertainty in comparisons to phytotoxicity thresholds
  - Uncertainty in risk characterization

- Uncertainties related to the wildlife assessment
  - Wildlife exposure estimates
  - Reference exposure estimates
  - Representativeness of sampling locations
  - TRVs
  - Uncertainty in TRV extrapolation
  - Population-level uncertainty
  - Uncertainty in risk characterization.

These sources of uncertainty and their effects on risk characterization conclusions are discussed in the following sections.

6.6.1 Uncertainties Related to Reference Area Selection

This section describes the selection and use of the reference areas in the risk assessment, reviews uncertainties about the reference area data, and discusses implications of these uncertainties for the use of the reference area data and the findings of the risk assessment.

6.6.1.1 Terrestrial Reference Area

Terrestrial reference areas were selected after review of existing studies and data, with a focus on factors such as prevailing wind directions, bedrock geology, topography and physiography (including slope, aspect, and water features such as streams and tundra ponds), and plant and animal communities. Possible reference areas were considered to the east, north, west, and south of the mine and DMTS. The prevailing wind originates from the east, between the northeast and southeast quadrants; thus, the most significant dust deposition has occurred to the north and west of the DMTS road and mine. As a result, areas to the north and west were not preferred areas for establishing the terrestrial reference area. Areas to the east were eliminated because the topography is more mountainous than most of the DMTS area. Thus, the focus was on selecting an area to the south of the mine and DMTS road. However, selecting an area too far south would have put the reference area into the Noatak valley, where the plant community includes trees and would not be as good for comparison with plant communities at the site. Therefore, the terrestrial reference area was targeted for placement somewhere within several miles south of the DMTS. Within that band south of the DMTS, the selected area was to be in a geologic area known to be relatively free of lead/zinc base metal mineralization. The selected area also needed to contain a variety of topographic conditions (elevations, slopes, and aspects), streams and ponds, and plant communities, providing the opportunity to sample environments similar to those along the length of the DMTS road. Based on these criteria, the Evaingiknuk Creek drainage was selected as the best choice. This basin met the most criteria, and had low base metal mineralization compared with other possible reference locations that were considered to the south of the DMTS.

Subsequent to the selection of the Evaingiknuk Creek drainage as the terrestrial reference area, sampling was conducted in two phases. The first phase included sampling of moss, which, when included with the overall moss database (including the NPS data, Ford and Hasselbach 2001, Hasselbach 2003, pers. comm., Hasselbach et al. 2005) and plotted together, provided a clearer perspective on overall patterns of deposition in the areas surrounding the DMTS and mine (Figure 1-9). Prior to the first phase of sampling, no moss data were available in that area.

The mean lead concentration for the three moss samples in the reference area is 8.0 mg/kg. Tundra soil was also sampled in the reference area, and the lead concentration ranged from 2.9 to 23.3 mg/kg, with a mean of 8.9 mg/kg, very similar to the mean moss lead concentration. In the area beyond approximately 16 miles north of the DMTS, where there is no apparent trend in the NPS moss concentration data, the mean lead concentration in moss is 8.5 mg/kg, or 6.4 if one outlier duplicate sample is excluded (Dixon's outlier test was used to confirm that the 38.6 ppm lead result is a statistical outlier at the 0.05 level [$0.02 < P < 0.05$]). The concentrations in the reference area and the area beyond 16 miles north of the DMTS appear to be similar. In the southern extent of the CAKR, beyond 12 to 13 miles south of the DMTS, the NPS moss lead concentrations average 2.0 mg/kg. It should also be noted that the area surrounding the Red Dog district is more mineralized than the southern part of CAKR. If there were dust depositional influence in the reference area, or the northern extent of the data collection area, it would appear to be very limited.

The communities in the reference area appear to be healthy, unimpaired communities suitable for use in reference/site comparisons. Even if there were some evidence suggesting low-level deposition in the reference area, the potential for this dust deposition to cause adverse effects to receptors is minimal. The metals concentrations in moss and lichens were very low; copper and zinc concentrations were far below effects levels reported in the literature (e.g., see Tables 6-19 and 6-20 for moss and lichen comparisons with threshold values). Furthermore, in almost every case, metals concentrations in terrestrial sedge and shrub samples were below phytotoxicity thresholds, even though samples consisted of unwashed tissues (Tables 6-17 and 6-18). Lead and zinc exposures for all wildlife receptors were uniformly low and never exceeded TRVs in the terrestrial reference area. Hazard quotients did exceed 1.0 for some receptors in the reference area, particularly for aluminum and barium, although as discussed in the risk assessment, this appears to be a function of the conservative nature of the TRVs for these metals rather than their concentrations in reference area media. For example, aluminum concentrations in reference area moss were similar to or less than concentrations in the southern extent of the CAKR, many miles further away in a prevailing upwind direction from the DMTS. This would suggest a similar level of risk would be predicted from aluminum in the southern part of the CAKR. However, because that area is well beyond the potential influence of the DMTS, it just illustrates the overly conservative nature of the aluminum TRV.

6.6.1.2 Coastal Plain Reference Area

In the second phase of sampling, a plant community assessment was conducted, and in order to better match the coastal plain plant community at the port, an additional reference area was selected south of the port in the CAKR (sample station TS-REF-12). Although moss was not collected at this location, tundra soil had a lead concentration of 5.8 mg/kg, slightly lower than the 8.9 mg/kg concentration in the terrestrial reference area.

6.6.1.3 Reference Lagoons

The reference lagoons included the Control Lagoon, approximately 2 miles south of the port, and an unnamed lagoon approximately 5 miles south of the port. The Control Lagoon was established as a reference in early port site studies (ENSR 1990), and the unnamed “Reference” lagoon was added during the first phase of the risk assessment sampling efforts (Exponent 2003e). At these distances, any depositional influence would be small, given prevailing wind directions. Mean sediment concentrations (from the 2003 and 2004 sampling events) in the two lagoons at different distances from the site are almost identical, with lead 9.6 and 9.5 mg/kg, zinc 86.6 and 86.9 mg/kg, and cadmium 0.2 and 0.3 mg/kg in the Control and Reference lagoons, respectively.

6.6.1.4 Marine Reference Area

The marine reference area is located approximately 3 miles to the south of the port. Sediment samples were collected there during several marine sampling events. Even if there were any depositional influence this far south, the influence would be very slight, and would likely be largely dissipated by dynamic ocean action, including wind, waves, and prevailing northward currents. Regardless of whether there is any detectable influence at the marine reference area, site sediment data from recent sampling events have been below all available screening thresholds, as described in Section 4.3.

6.6.1.5 Effect of Uncertainties

There are clearly uncertainties with regard to the potential influence from dust deposition on reference areas. However, the possible effect of these uncertainties on the analyses, such as comparison of site and reference area conditions, appears to be limited. Based on the discussion in Section 6.6.1.1, there is very little if any measurable depositional influence from the mine within the terrestrial reference area. Thus, the possible influence of mine dust deposition in the reference area is so small as to be highly unlikely to result in any incremental effects to receptors in that area. Therefore, comparisons of communities (e.g., benthic and plant communities) at the site with those in the reference area are acceptable for the analyses. Further discussion of uncertainty related to the use of reference area comparisons in CoPC selection is included below in Section 6.6.3.

6.6.1.6 Summary

While all of the reference areas are suitable for the risk assessment, there are clearly some uncertainties with regard to the potential influence from dust deposition. The possible need for additional study to further address these uncertainties will be considered during development of a risk management plan.

6.6.2 Uncertainties Related to Sample Size

The knowledge gained from collection of data in multiple media results in a fairly clear picture of depositional patterns associated with site sources. The conceptual model lends itself to interpreting smaller data sets within a given medium, whereas in the absence of the broader conceptual model, those smaller data sets may be of less value. The small samples sizes are the result of balancing resources and available time to collect data for the risk assessment during two short field seasons. The risk assessment provides an understanding of what the potential risks are, thereby helping to focus additional information gathering on receptors and/or media where these potential risks were identified. Thus, the limitations of the small sample sizes can be further addressed, as needed, in future monitoring focused on areas of potential risk identified as part of this assessment.

6.6.2.1 Tundra Ponds

The field sampling plan was developed to include multiple tundra pond sediment and surface water stations. However, extensive searches by field staff located fewer tundra ponds than the number originally planned to be sampled. Therefore, the number of samples is limited because of the spatial distribution and availability of tundra ponds.

6.6.3 Uncertainties Related to CoPC Screening

Two screening approaches were used to identify CoPCs for ecological receptors. In the first approach, to select CoPCs for plants, invertebrates, and fishes, maximum chemical concentrations in tundra soil, sediment, and surface water in different environments at the site were compared against ecological screening benchmarks and then statistically compared against reference area concentrations. As in the human health CoPC screening, those chemicals that both exceeded their ecological screening benchmark *and* were significantly different from reference area concentrations were retained as CoPCs. Chemicals without screening benchmarks were evaluated based on reference comparisons alone; if chemical concentrations at the site were statistically significantly different from those at the reference area, or if statistical comparisons could not be made, the chemical was retained as a CoPC in that environment. The following chemicals were eliminated from further consideration in the baseline ERA because site concentrations were not statistically significantly higher than reference concentrations:

- Tundra soil: aluminum, chromium, iron, and nickel
- Stream sediment: barium and vanadium
- Stream water: aluminum, cobalt, iron, and strontium
- Pond sediment: barium, copper, nickel, selenium, thallium, and vanadium
- Pond water: aluminum, barium, cobalt, copper, fluoride, iron, lead, molybdenum, strontium, and vanadium
- Lagoon sediment: aluminum, barium, cobalt, iron, molybdenum, nickel, selenium, strontium, thallium, and vanadium
- Lagoon water: aluminum, arsenic, barium, cobalt, iron, nickel, strontium, and zinc

- Marine sediment: none (site concentrations were below the SQS)
- Marine water: aluminum, antimony, barium, cobalt, fluoride, iron, manganese, and molybdenum.

In the second approach, 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, including herbivores foraging in the tundra (tundra vole), piscivores foraging in streams (red-throated loon and river otter), and invertivores foraging in streams and ponds (common snipe) and coastal lagoons (black-bellied plover). 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. Chemicals were retained as CoPCs for wildlife when their exposures exceeded the TRVs (hazard quotients were greater than 1.0) *and* were at least twice as high as the corresponding reference exposures. The following chemicals were eliminated from further consideration in the baseline ERA, although their hazard quotients exceeded 1.0, because exposures at the site were less than twice as high as reference exposures:

- Tundra vole: manganese
- River otter: none (no reference data were available)
- Red-throated loon: none (no reference data were available)
- Common snipe
  - Streams: aluminum, barium, and chromium
  - Ponds: aluminum, barium, chromium, and selenium
- Black-bellied plover
  - Lagoons: aluminum, barium, chromium, and mercury.

The surface water, sediment, and soil benchmarks used in the screening evaluation are generic values developed using a variety of test species and experimental conditions that may not be representative of the receptors and site-specific environmental conditions. Therefore, application of these generic values adds uncertainty to the screening risk assessment because these values may not be directly relevant to environmental conditions at the site (e.g., acclimation of ecological receptors over time to site-specific factors, differences in bioavailability of CoPCs, heterogeneous sediment or soil matrices) that could potentially reduce the likelihood of adverse ecological effects even when CoPC concentrations in media exceed benchmark values. However, because screening is intended to be a conservative process designed to avoid elimination of chemicals that may present unacceptable risk to ecological receptors, the result of these uncertainties is most likely the retention of chemicals for subsequent evaluation that actually pose no unacceptable risk. Such chemicals are identified and eliminated in the baseline assessment when some of the assumptions used in screening are relaxed and replaced with site-specific data or more ecologically relevant exposure assumptions. Although the CoPC screening process was generally conservative, ecological screening benchmarks were not available for some of the components of the tundra ecosystem most vulnerable to metals deposition, i.e., mosses and lichens. It is not known if the ORNL toxicological benchmarks for vascular plants, earthworms, and soil fauna that were used to identify CoPCs in tundra soil were also protective of nonvascular plant species.

Sulfur was eliminated from the list of CoPCs for the ERA, because it is naturally abundant in the environment, and it is not on EPA's target analyte list, nor on DEC's list of hazardous substances for which cleanup levels are provided in 18 AAC 75.340 and 18 AAC 75.345 (see Section 3.1). Therefore, sulfur data were not collected in support of the risk assessment.

Sulfur is a plant nutrient, mostly taken up from the soil as sulfate, but also absorbed from the air as sulfur dioxide via stomata. The majority of sulfur dioxide toxicity literature is based on studies that have evaluated lichen as bioindicators of air pollution, because lichen are known to be very sensitive to phytotoxic effects of sulfur dioxide. Lichens (especially when moist) can become a large sink for sulfur dioxide because of the compound's high solubility in water. A study by Nash and Gries (2002) found that approximately 70 percent of the absorbed sulfur dioxide can be oxidized to sulfate and leached from lichens, which acts as a detoxifying mechanism. However, the authors also reported that retained sulfur dioxide can be converted to bisulfite, and can be toxic when accumulated at high levels as a result of acidification and necrosis of plant tissue. Nash and Gries (2002) explained that toxicity effects on lichen usually manifest as decreases in photosynthesis and respiration, leaching of electrolytes, spore generation, and increased mortality.

Studies on decreased photosynthesis effects included tests on lichen exposed to 170 $\mu g/m^3$ to 2,500 $\mu g/m^3$ sulfur dioxide in air (Richardson and Nieboer 1983). Decreases in spore germination and spore germination inhibition were found in lichen that were exposed to aqueous sulfur dioxide at concentrations of 0.032 mg/L (Belandria et al. 1989). In laboratory experiments, Grace (1981, as cited in Richardson and Nieboer 1983) found that lichen exposed to 14,600 $\mu g/m^3$ sulfur dioxide in air resulted in potassium leaching. Potassium efflux is interpreted as an increase in cell permeability. McCune (1988) observed that lichen community parameters (species composition, species richness for example) were correlated with 3-year mean annual sulfur dioxide levels ranging from 23 to 40 $\mu g/m^3$ in Indiana.

Liblik and Pensa (2001) observed that critical levels for sulfur dioxide range from 10–30 $\mu g/m^3$ for general vegetation, but for sensitive lichen and *Sphagnum* mosses, a critical limit of 3–9 $\mu g/m^3$ was suggested. Kashulina et al. (2003) summarized that the critical levels of sulfur dioxide emissions (in annual mean averages) are 15 $\mu g/m^3$ in air for trees growing in cold climates, and 10 $\mu g/m^3$ in air for the most sensitive plants, including moss and lichens. The authors, however, recommended that, based on moss damage in Kola Peninsula of arctic Russia, the critical level of sulfur dioxide in air for mosses and lichens should be set lower than previously proposed, to 5 $\mu g/m^3$.

The question of potential sulfur effects on lichens is an area of uncertainty. The need for future studies of nonvascular plants will be evaluated during development of the risk management plan.

Overall, the ERA identified potential risks from several of the CoPCs carried through the risk assessment, and/or from fugitive dust in general. For wildlife, risks are likely to be lower for chemicals that were not carried through the assessment than they are for the risk-driving chemicals that were evaluated in the assessment. For lichens and other tundra vegetation, the observed effects are probably the result of multiple constituents in fugitive dust, possibly including chemicals that were not evaluated individually in the baseline risk assessment, such as sulfur compounds, as well as chemicals that would typically be present in dust from any gravel road. During development of the risk management plan (discussed in Section 8.1), the risk assessment results can be used to prioritize future actions such as additional data collection, monitoring, and/or implementation of additional dust control. If there are future changes in site concentrations of metals that were screened out by comparison with reference areas, and those changes are related to fugitive dust deposition, the changes will be detected by concomitant changes in the concentrations of CoPCs that are included in future monitoring programs. In addition, future monitoring of tundra vegetation will identify changes in community conditions resulting from changes in the concentrations of all dust-related chemicals, whether or not those chemicals are specifically analyzed for in future monitoring studies.

6.6.4 Uncertainties Related to the Terrestrial Assessment

Important sources of uncertainty in the terrestrial plant assessment include the following:

- Uncertainty in the plant community surveys
  - Representativeness of sampling locations
  - Selection of reference sites
  - Field sampling methods
  - Sample size
- Uncertainty in comparisons to phytotoxicity thresholds
- Uncertainty in risk characterization.

Uncertainties associated with the plant community surveys are described in Section 6.6.4.1, followed by a discussion of the uncertainties surrounding phytotoxicity threshold comparisons (Section 6.6.4.2). Uncertainty in the risk characterization for terrestrial plants is discussed below in Section 6.6.4.3.

6.6.4.1 Uncertainty in the Plant Community Surveys

The primary sources of uncertainty associated with the plant community surveys relate to aspects of data collection, including sampling station locations, reference site selections, field methodology, and sample size, as discussed in detail in the subsections below. Coastal lagoon plant communities were also characterized using the same survey techniques as the terrestrial vegetation assessment (see Section 6.2). Therefore, most aspects of the following discussions are also relevant for the interpretation of lagoon community survey results.

Another uncertainty is that conditions currently experienced by plant communities may change over time as a result of weathering of metals-bearing fugitive dust in the tundra environment. It is possible that some metals may become more available to plants as weathering occurs. Further study would be required to evaluate this possibility.

6.6.4.1.1 Representativeness of Sampling Locations

Plant community survey transects were distributed along the length of the DMTS road in order to evaluate plant communities that are exposed to varying levels of site-related CoPCs, with higher exposures generally occurring at each end of the road, near dust sources at the port facilities and mine, and lower exposures generally occurring in the central portion of the DMTS road. Major plant communities included coastal plain mesic tussock tundra, foothills mesic tussock tundra, and hillslope mesic open shrubland. Within each transect, vegetation surveys were conducted on survey lines parallel to the DMTS road at several distances from the road to assess community changes with distance from dust sources. Transects were all oriented roughly to the north, predominantly downwind of the DMTS road, which was a conservative sampling design. Thus, the plant community surveys evaluated vegetation that was representative of major terrestrial plant communities in the DMTS road corridor, and addressed areas most likely to be influenced by fugitive dust. Some communities around the mine’s ambient air/solid waste permit boundary, such as dry alpine tundra communities, were not evaluated quantitatively in the risk assessment, and risk assessment results cannot be extrapolated to these distinct vegetation communities. However, permanent vegetation monitoring quadrats have been established in three plant community types in the mine area (ridgetop dwarf shrub tundra, dwarf birch and blueberry shrub community, and tall willow community; RWJ 1998), and four monitoring quadrats were established in a reference area located 3.6 miles southeast of the mine’s Personnel Accommodation Complex (RWJ 1998). One reference quadrat was established in ridgetop and birch/blueberry communities, and two reference quadrats were established in the tall willow community. Therefore, any changes to these communities would become apparent as a result of this periodic monitoring. However, please note that the area inside the mine boundary is beyond the scope of the DMTS risk assessment.

Terrestrial plant community surveys were limited to stations up to 1,000 m from the DMTS road or 1,430 m from the DMTS port facilities (station TT5-2000). For some endpoints, such as average lichen cover, there are indications that road effects may still be present at these distances. Therefore, the lack of survey data beyond 1,000 m from the DMTS road limits the study’s ability to completely delineate the full extent of discernable changes for a few characteristics of plant communities.

6.6.4.1.2 Selection of Reference Stations

Reference stations for tundra and hillslope communities were located south (predominantly upwind) of the DMTS road corridor, in the Noatak River drainage. The geology of this area is less mineralized than the site, and therefore it is a conservative reference against which to compare CoPC concentrations in environmental media and biota, or modeled CoPC exposures for wildlife. However, there are some differences between the DMTS road corridor and the terrestrial reference area that may affect their respective plant communities. Stations along the DMTS road are in the Wulik River drainage, which flows to the northwest. In contrast, the Noatak River drainage flows to the southwest. The terrestrial reference area is also topographically more complex than the majority of the DMTS road corridor, resulting in potential differences in wind and water drainage patterns that could influence vegetation communities. Differences in drainage patterns made it challenging to find reference stations with comparable slope and aspect to the site stations. For example, reference station TS-REF-5 was located near the crest of a low rise, whereas site stations were situated downgradient of a physical barrier, the DMTS road. Tundra reference stations were also closer to treed areas than site stations; alder and spruce trees were observed near hillslope reference station TS-REF-11, suggesting that the reference area was near a vegetation transition zone. Additionally, field observations for environmental factors such as topography suggested that tundra transects at the site may be more similar in some aspects to the coastal plain environment (i.e., transect TT5 and coastal plain reference station TS-REF-12) than to the foothills tundra community seen at the tundra reference stations. In the plant community assessment (Section 6.2), coastal plain and tundra communities were evaluated separately and in combination, in order to distinguish trends unique to each community and also to identify any consistent trends across the site. In light of the uncertainties associated with reference station selections, comparisons between site and reference communities were evaluated together with other lines of evidence in this risk assessment, and the weight of the evidence used to characterize the potential for adverse effects to terrestrial vegetation.

6.6.4.1.3 Field Sampling Methods

Field sampling methods were applied consistently at all site and reference stations, so that any uncertainties in the data collection should be reflected consistently in both site and reference results. One observer performed all the plant community surveys, and therefore any observer bias should also be consistent across stations.

Percent cover for vascular plant species was estimated in two dimensions, such that any canopy area that occurred under the cover of another, taller plant species was not included in the cover estimate. Thus, understory plants such as some forbs or prostrate shrubs could occur in microplots but not contribute to the canopy cover, because taller plants such as deciduous shrubs were shading them. Therefore, changes in cover for understory plants may be a result of changes in abundance of that species or plant functional group, or may simply reflect changes in a higher canopy. Therefore, both cover and frequency estimates were used in the interpretation of plant community data. Percent frequencies of plant species were used to determine whether species are present in a community and to gauge how common they were.

Species diversity and evenness calculations were based on cover estimates, so only those species that contributed to total canopy cover were included. The exclusion of species occurring at trace cover levels biased diversity indices lower and evenness indices higher, and would have a greater effect on communities with many rare species or where taller shrubs created canopies over other plants. The uncertainties associated with two-dimensional cover estimates are not relevant for moss and lichen covers, which were estimated independently of the vascular plant canopy coverage. Cover percentages for these groups (and other categories such as bare ground) were total estimated covers for the whole microplot.

Cover percentages were estimated in each microplot using cover classes (cover ranges), rather than attempting to assign discrete cover values to plants, which could imply an unwarranted degree of precision given the spatial and temporal heterogeneity of plant communities in nature and potential inconsistencies of visual estimates (Barbour et al. 1980). Cover class midpoints were used to calculate average species covers over the ten microplots, and the midpoint may either overestimate or underestimate the actual cover in each microplot. However, average cover percentages should reasonably reflect species' covers. Because some cover classes equate to fairly broad ranges (e.g., 25–50 percent), this method may not detect subtle changes in cover, such as those that occur within a cover class's range. Nonetheless, uncertainty associated with cover class estimates is not likely to affect risk conclusions for plant communities.

Timing of the field event may have affected cover, frequency, and richness measurements. Plant community surveys took place over the course of a month, during which time many plants began to flower or, alternatively, finished flowering and went to seed. Thus, some plant species such as grasses lacked distinguishing characteristics early in the field program but were more readily identifiable later in the season. The field notes indicate that, based on the results of the survey at reference station TS-REF-12, which was sampled late in the program, some grass species may have been missed in the characterizations of coastal plain plant communities at TT5-1000 and TT5-2000, which were sampled early in the program. However, great attention to detail was placed on species identification. Vouchers were retained, and those relative few for which identification was uncertain were reassessed during the course of the field program to confirm results. Overall, the effect of uncertainty associated with the timing of the surveys on survey results is expected to be insignificant.

Moss and lichen communities were evaluated only to group level rather than to the species level. Thus, potential changes in moss or lichen community compositions related to fugitive dust from the DMTS road cannot be assessed. Species-level information is not needed to identify stress to mosses or lichens as a whole but would help distinguish potential effects to more sensitive members of these communities.

Despite the various sources of uncertainty in the plant community surveys, the data did reveal trends in plant communities that suggest an influence from the DMTS road. Although it is important to consider the limitations of the sampling design and the uncertainties associated with the sampling methods when interpreting the plant community survey results, the uncertainties described above do not affect the overall risk conclusions for terrestrial plants (Section 6.2.3).

6.6.4.1.4 Sample Size

To explore whether a sufficient number of microplots was evaluated to adequately characterize the vegetation at a given station, the cumulative number of vascular plant species identified at a station was plotted over the total area surveyed (up to 10 m^2, equivalent to the area inside 10 microplots) as shown in Figure 6-23. Each graph summarizes the results for one plant community type (e.g., coastal plain mesic tussock tundra), and each curve on the graph shows the cumulative number of vascular plant species identified in successive 1-m^2 microplots assessed at a given station (e.g., TT5-0010). The data are plotted along the x-axis in the order in which the microplots were evaluated in the field.

Species-area curves for the coastal plain community suggest that ten microplots were sufficient to capture most species (Figure 6-23). In fact, no new species were identified after the fifth microplot at station TT5-2000, after the sixth microplot at station TT5-1000, and after the eighth microplot at stations TT5-0100 and TT5-0010. Similarly, the species-area curves for stations on tundra transects TT3 and TT8 seemed to plateau, with few new species added with increasing area. Coastal lagoon communities had low species richness compared to terrestrial plant communities, and most or all species were identified in the first few microplots examined. At hillslope stations, however, the species-area curves suggest that ten microplots were not adequate to characterize these diverse communities.

Based on the discrepancies between species richness and area richness estimates (summarized in Table 6-15), it appears that ten microplots may not always have been sufficient to capture all the species observed at a survey station, particularly in disturbed sites near the road and port facilities and in the diverse hillslope community. Species that were observed in the general vicinity of the survey line but were not captured in microplots included forbs at station TT5-0010 (e.g., lousewort and buttercup), primarily shrubs at station TT5-0100 (e.g., blueberry and Labrador tea), and forbs, grasses, and willows at stations TT3-0010, TT3-0100, TT8-0010, and TT8-0100 (e.g., polar grass and bog willow). Plants are not evenly distributed in nature, and richness estimates based on microplot counts may miss rare species or species with patchy distributions. Species richness estimates were used in statistical calculations, because they were standardized measures and therefore comparable across stations. However, the approximate area richness estimates show that species richness values underestimate the number of species present in the community. While this uncertainty does not alter overall trends in species richness with distance from the road, it does affect site and reference community comparisons in a few cases. For example, based on species richness, hillslope stations TT6-0010 and TT6-0100 appear to have about the same number of species as the reference station, TS-REF-11 (25 and 23, respectively, as compared to 24; Table 6-15). However, based on area richness estimates, the site stations have lower species richness than the reference station (29 species at either site station, as compared to 35 species at the reference station; Table 6-15). Likewise, lagoon station PLNL appears to be less rich than reference station CL-REF-1 based on the species richness values but is actually more species-rich based on the area richness values (Table 6-15).

Natural variability in tundra communities may obscure differences related to fugitive dust effects, given the small number of replicates in this study. Plant communities along the DMTS shifted in response to changes in topography, drainage, aspect, elevation, local geology, or other environmental factors. The single coastal plain transect at the port and two tundra transects in the central portion of the road were distributed many miles apart, where elevation changes and other environmental factors likely influenced vegetation patterns to some degree. No replicates were sampled in the hillslope community near the mine. The three reference stations evaluated in coastal plain and foothills mesic tussock tundra environments may not have been sufficient to account fully for natural variation in characteristics, such as the relative dominance of plant functional groups or the commonness of individual species. Only one reference station was evaluated in the hillslope mesic open shrubland community, and two reference coastal lagoons were surveyed. Because of small sample sizes, real differences between site and reference communities or differences in communities with distance from the DMTS road may not have been detected in statistical tests; therefore, a less stringent p-value of 0.10 was used in the tests to compensate for the low number of samples. Thus, comparisons between site and reference stations must be approached with caution, and in this risk evaluation were interpreted in context with other lines of evidence, such as trends with distance from the road and port facilities.

6.6.4.2 Uncertainty in Comparisons to Phytotoxicity Thresholds

Tissue concentrations of CoPCs in terrestrial vascular and nonvascular plants collected from site and reference stations were compared against phytotoxicity threshold values reported in the scientific literature. These comparisons were also assessed for plants in freshwater aquatic environments and along the margins of the coastal lagoons. The phytotoxicity thresholds for vascular plants are generic values based on studies of agricultural crops or other plants that are not specific to arctic environments. These values may overestimate or underestimate potential effects to plants at the site, depending on the relative metals sensitivities of the site and test species. Tolerant varieties of higher plants are known to occur in areas of natural mineralization or anthropogenic deposition and may accumulate high metals levels in their tissues without phytotoxicity (Kabata-Pendias and Pendias 1992). However, the specific tolerance or sensitivity of various plant populations in the DMTS road corridor is not known.

Phytotoxicity thresholds for mosses and lichens are derived from a field study of a coniferous woodland community near a brass foundry (Folkeson and Andersson-Bringmark 1987). Although the woodland plant community obviously differs in composition from the tussock and shrub tundra communities examined in this risk assessment, the phytotoxicity thresholds reported in the study were derived for mosses and lichens that are dominant in the tundra communities along the DMTS, and for which there are site-specific chemical data (e.g., feather mosses and reindeer lichens). Therefore, these threshold values are directly relevant to common cryptogams exposed to fugitive dust at the site. This study was also selected for the following characteristics: realistic field setting and chronic exposure duration comparable to years of dust deposition along the DMTS, as opposed to acute laboratory toxicity tests; relevant CoPCs, especially zinc; corresponding concentration and response gradients; comparable tissue data (metals concentrations in unwashed samples); and ecologically relevant endpoints, including survival and abundance (cover). However, because the moss and lichen toxicity thresholds are not site-specific, and are based on a single study from the literature, there is uncertainty associated with their application to the plant communities along the DMTS, where conditions inevitably differ somewhat from the test site. For example, copper concentrations in soils near the foundry are much higher than concentrations found along the DMTS, whereas zinc concentrations are similar at the two sites. Also, soil pH changes and other factors related to foundry emissions or environmental conditions may confound the relationships between observed effects and metals concentrations in tissues reported in the study.

Concentrations of CoPCs in both sedges and shrub leaves (willow and birch) were compared against threshold values to evaluate the likelihood of adverse effects to different components of terrestrial plant communities. Metals concentrations were not available for other plants, such as forbs or evergreen shrubs, and no comparisons could be made for these plant groups. Therefore, there is uncertainty in the extrapolation of results for individual species to the plant community as a whole, as other species may respond differently to CoPC exposures.

Plant samples from the site were not washed before analysis, and thus CoPC concentrations for these samples include metals on plant surfaces as well as metals inside plant tissues, the relative proportions of which cannot be determined from this study. For samples collected near the DMTS road or port facilities, where dust deposition is highest, surface metals may compose a substantial fraction of the total, and sample concentrations may overestimate the plant's uptake of CoPCs. When compared to literature values derived from controlled uptake studies, these concentrations may overestimate the potential for adverse effects, giving a conservative bias to the phytotoxicity screening results. Phytotoxicity values may be derived based on exposure to solutions of metals, which may overestimate the availability to and uptake by plants of the relatively insoluble forms found in road dust and soil.

Phytotoxicity threshold comparisons are inherently very uncertain. Threshold values are chemical-specific and may not account for possible additive or antagonistic effects of exposures to multiple CoPCs, or additive effects resulting from non-chemical stressors. Generic literature values may not be appropriate for comparison with arctic species or individuals of potentially tolerant populations that evolved in highly mineralized areas, particularly in the vicinity of Red Dog Mine. Therefore, comparisons with literature values were used as a supplemental line of evidence to community evaluations in the risk characterization for terrestrial plants (Section 6.2.3).

6.6.4.3 Uncertainty in Risk Characterization

Multiple lines of evidence were considered in the risk characterization for terrestrial plants, including site and reference comparisons, relationships with distance from the DMTS road, correlations of vegetations and tundra soil parameters, PCA and NMDS trends, qualitative assessments of plant vitality, and comparisons between plant tissue concentrations and phytotoxicity thresholds. The use of multiple indicators to evaluate potential effects to terrestrial plants enhances confidence that site-related changes in vegetation communities have been identified and that the alterations are related to the influence of the DMTS road.

The causes of vegetation effects are not known, because tundra soil parameters such as CoPC concentrations and pH are significantly correlated with distance from the road (Table 6-4), and these or other physical factors may potentially contribute to the changes in vegetation communities near the road. It is difficult to determine the relative significance of physical and chemical factors in the vegetation effects observed, because both are correlated with distance from the road. Tundra soil parameters, such as CoPC concentrations and pH, are significantly correlated with distance from the road (Table 6-4), thus, these as well as other physical factors, may potentially contribute to the changes in vegetation communities near the road. Physical factors are likely to exert their greatest influence near the road and port facilities where dust deposition is greatest and drainage may be locally altered. Chemical factors (elevated metals and pH) are likely to become more important than physical factors at greater distances from dust sources, but are also likely to be a significant factor in changes observed near the road and port. Studies of dust deposition along the Dalton Highway have shown that the majority of dust is deposited within 500 m of the road or less. Lamprecht and Graber (1996) modeled fugitive dust deposition along the Dalton Highway and predicted that 20–45 percent of the dust would settle out within 40 m of the road; 65–95 percent would settle out within 200 m of the road; and 75−98 percent would settle out within 400 m of the road. The authors "conclude that at any location along the Dalton Highway, road dust emitted by truck movement should settle out to 98 percent within an area of less than 500 m of either side of the road." Walker and Everett (1987) measured dust loads along the Dalton Highway using dust collection pans and found that 97 percent of the dust was deposited within 125 m of the road, although silt and clay-sized particles were deposited up to 1 km or farther from the road.

If dispersion were strictly a function of particle size, concentrate dust would be expected to travel farther than coarse roadbed material (i.e., sand and gravel) but would be expected to behave similarly to the fine particles in road dust. The most common size fraction of dust particles collected over 24 hours at locations 30 m, 70 m, 150 m, and 300 m from the Dalton Highway was the 10–20 μm diameter range (Lamprecht and Graber 1996). Walker and Everett (1987) observed a decrease in median particle size with distance from the road, from predominantly 0.5–2 mm particles at the road source to 0.02–0.25 mm particles at 8 m from the road, to 2–50 μm particles at 125 m and 312 m from the road. The particle size of zinc and lead concentrates is <40 μm, with 80 percent <20 μm (Teck Cominco 2003b,f).

Vegetation effects along the Dalton Highway tended to coincide with dust deposition and were most pronounced in areas of heavy dust close to the road. Auerbach et al. (1997) assessed vegetation characteristics up to 800 m from the Dalton Highway and observed the greatest effects within 100 m of the road. The 400-m and 800-m samples "were predicted as being beyond the extent of major dust effects." However, the authors did not survey vegetation beyond 800 m. Auerbach et al. (1997) reported that r^2 results of regressing lichen species cover on log of distance from the Dalton Highway ranged from 0.13 to 0.42, and r^2 results of regressing lichen biomass on log of distance from the Dalton Highway was equal to 0.018 and 0.28. These r^2 values reported by Auerbach et al. (1997) are lower than results for lichen cover on coastal plain and tundra transects along the DMTS road (0.77, Table 6-4), potentially reflecting the additional contribution of metals toxicity from fugitive concentrate dust to lichen declines near the DMTS. However, the studies are not directly comparable because of sample size differences: Auerbach et al. (1997) sampled eight stations with five replicate plots for a total of 40 samples per site, compared with 10 samples in the combined coastal plain and tundra communities. The differences in sample sizes could influence the magnitude of the r^2 values in the two studies.

Walker and Everett (1987) noted the most extreme vegetation effects (e.g., elimination of mosses) within 10 or 20 m of the highway, while effects to lichen communities extended beyond 70 m. The authors focused their report on vegetation effects in heavy dust areas and did not investigate potential plant community changes at distances beyond 100 m.

Similar to the Dalton Highway studies, vegetation effects along the DMTS road corridor were also most pronounced near dust sources; however, results for lichens suggest that effects may extend beyond distances at which communities were altered along the Dalton Highway. Further study would be required to elucidate the role and spatial gradient effects from site-related CoPCs relative to other road effects commonly observed elsewhere in Alaska.

As stated in Section 6.4.2.4, *Risk Characterization of Coastal Lagoon Plants*, fringing coastal lagoon vegetation does not appear to be adversely affected by exposures to CoPCs at the site. Plant tissue concentrations did not exceed phytotoxicity thresholds, and qualitative assessments of vegetation vitality did not reveal any evident signs of phytotoxicity. Other plant communities occur in the vicinity of the coastal lagoons, some of which may be more sensitive to metals deposition than wetland graminoid communities. These different communities were not surveyed and are not directly evaluated in the risk assessment, and extrapolation of the results for fringing wetland vegetation to other coastal lagoon communities may not be appropriate.

6.6.5 Uncertainties Related to the Wildlife Assessment

The risk characterization for wildlife is based on an individual-based model that is intended to predict the response of a population of wildlife receptors as the result of the presence of a number of potential toxicants (i.e., the CoPCs) in a particular location, at a particular concentration, at a particular time. Through the development of multiple station- and assessment unit-based risk scenarios, the risk characterization takes into account the distribution of CoPCs at the site and combines this information with estimated values for key life-history parameters of the receptors and predicted physiological responses to CoPC exposure to provide a measure of the likelihood that the conditions, as understood, will affect receptor population demography. The risk assessment is, however, only a model of reality. By virtue of incomplete knowledge on receptor ecology and toxicology, models must generalize over conditions, assume events and responses, and disregard factors and conditions based on the presumption that such factors are inconsequential. Best professional judgment is applied to ensure that while the models do not significantly underestimate the potential risks, they do not become so conservative as to render the results meaningless. The specific uncertainties associated with the risk assessment for wildlife are identified and discussed in the following sections.

6.6.5.1 Wildlife Exposure Estimates

Exposure estimates for wildlife receptors were based on a model that incorporated site-specific data on CoPC concentrations in food and environmental media with assumptions about the life history characteristics of the receptor species. The food-web exposure analyses were deterministic and incorporated mean or 95%UCL on the mean CoPC concentrations and receptor-specific exposure parameters. Almost all of these values have associated probability distributions; however, selection of determinate values for the exposure and effects characterizations was based on the best available information on the average individual. Where possible (generally for large-home-range receptor scenarios), 95%UCLs were used. Point values were used for station-based estimates for small-home-range receptors. In the absence of site-specific information on parameters such as body weights, prey selection, and ingestion rates, information was obtained from literature sources. Uncertainty is inherent in all the assumptions used to estimate the exposure of receptors to CoPCs in the DMTS road corridor. However, these assumptions are as ecologically accurate and realistic as possible. Where uncertainty was identified, values were selected that would tend to maximize exposure or effect and therefore would be conservative in the estimation of risk. Below is a detailed discussion on specific sources of potential uncertainty that have been identified in the food-web exposure models.

6.6.5.1.1 Body Masses and Intake Rate Parameters

The application of single determinate values for exposure parameters introduces a level of error to the exposure estimates, because none of the parameters are constant to an individual all of the time or constant across individuals within a population. Body mass estimates were based on values reported in the scientific literature, with a focus on mean female body masses from Alaska or other northern regions (Table 6-30). Female body masses are used because most of the endpoints used as to establish NOAELs or LOAELs relate to reproductive parameters. Therefore, female exposure to CoPCs is important when predicting if population effects are likely to occur.

For many receptors, average male body mass may be higher than that of females, but because food ingestion rates scale with body weight and heavier organisms tend to eat proportionally less per unit mass, use of female data is not considered to underestimate effects to males. Food intake rates were taken from published observations or were calculated from mean body masses using allometric equations from Nagy et al. (1999; Table 6-30). Because lower weight individuals of a species eat at a proportionally higher rate than larger co-specifics, if a smaller body weight is used as an input parameter, then a higher ingestion rate per unit body mass is expected, which would increase the total exposure. For example, according to a comprehensive collection of mammalian body masses from throughout the world (Silva and Downing 1995), the lowest reported body mass of a tundra vole was 19 grams for voles in Poland. Using this body weight in place of the value of 47 grams used in this ERA, the food ingestion rate would be equal to 4.46 grams/day, compared to a rate of 8.5 grams/day used in this ERA. Therefore, the resulting hazard quotients would increase by a factor of approximately 1.5 from values stated in this report. However, because many species show variations in body size across their range, body mass data for populations in other locations, such as tundra voles in Poland, may not be representative of receptors present at Red Dog. Therefore, whenever available, life history information from arctic Alaska, or the next closest location, was used to select or derive exposure parameters, such as body weights, food ingestion rates, and diet compositions. Furthermore, mean values were selected to show the exposure, and hence risk, of the typical individual in a receptor population, with the understanding that this underestimates risk for some individuals in the population, yet also overestimates risk for other individuals.

Water intake rates were calculated using equations from U.S. EPA (1993) and were also based on mean body masses; water intake rates tended to be conservative, because they did not account for other sources of water such as moisture in prey and metabolic water, which can be significant. Thus, food-web exposure model results were representative of the average individual in a receptor's population and would tend to overestimate exposure for larger than average individuals and to underestimate exposure for smaller than average individuals.

Although there is a minor, non-quantified, exposure to wildlife via inhalation because receptors can be exposed to metals through incidental inhalation of fugitive dusts, this was considered to be a minor pathway for three reasons. First, the total exposure to metals in dust was considered to be small relative to the exposure received via ingestion of food and soil/sediment (U.S. EPA 1993). Second, relatively little inhaled dust is likely to pass into the lower respiratory tract and lungs, where absorption could potentially occur. Instead, most inhaled dust will likely end up being ingested (U.S. EPA 2003e). Third, metals would be bound tightly on dust particles and not readily available for uptake, unlike other chemicals, such as volatile organic compounds, which could be readily absorbed into the circulatory system from the lungs (U.S. EPA 1993).

U.S. EPA (2003e) has provided example calculations for the meadow vole, which allows for a comparison of percent contribution of the various pathways of exposure. According to their example, the percent contribution of particulates from the inhalation pathway is very low at less than 0.001 percent, while in contrast, the combined diet and soil ingestion pathways contribute more than 99.9 percent to the relative dose.

As noted in U.S. EPA (1993), calculation of dose deposited, retained, and absorbed in the respiratory tract is a function of many factors, including species anatomy, physiology, particle size distribution, and pharmacokinetic data. To accurately calculate the importance of the inhalation pathway would require use of physiologically-based pharmacokinetic models. However, these models only exist for a few common laboratory species and extrapolation to wildlife receptors would introduce considerable uncertainty to risk estimates that is disproportionate to the relatively low importance of this exposure pathway.

6.6.5.1.2 Diet Composition

The diet composition for each receptor was approximated using best professional judgment based on information found in the literature (Table 6-30). Because receptors were selected to represent feeding guilds (e.g., tundra shrew for terrestrial mammalian invertivores), their modeled diets emphasized primary food sources (e.g., invertebrates for tundra shrew). Use of multiple feeding guilds minimizes the likelihood that risk for any particular guild is underestimated. For example, insect matter constitutes a minor proportion of the vole diet. If insects have higher CoPC concentrations than plant matter, then omitting this component of the diet could underestimate risk for herbivorous small mammals that eat some animal tissue. However, the risk estimates for shrews would be protective of these receptors. The most appropriate tissue data collected in the field was used to represent food concentrations in the models. Diets were simplified for the purpose of the risk assessment, and because exposure estimates were determinate, they did not capture the temporal and individual variability in receptors' diets. Therefore, the simplification of receptors' diets introduced some uncertainty into the risk calculations.

6.6.5.1.3 Time Use

Because the caribou and many of the avian receptors selected for this risk assessment are migratory, it would have been unrealistic to assume that they were exposed to CoPCs from the DMTS for the entire year. Therefore, the assessment standardized exposure rates over an annual cycle and apportioned exposure concentrations from the site based on the proportion of time receptors are assumed to spend in that habitat. The selection of residence times for birds was generally not site-specific, but rather based on the typical times of the year that the specific receptor is most likely to arrive at and depart from Cape Thompson, Alaska (Table 6-30). In cases where a range of values was available, the period that maximized their residency at the site was selected, thus maximizing exposure to site CoPCs.

In most years, the majority of caribou in the WACH migrates to river drainages south of the site in autumn and does not over-winter in the DMTS road corridor (see Section 6.1.6.1). These animals would be exposed to CoPC concentrations at the site for short durations rather than 5 months as modeled (Table 6-30). As discussed above in Section 6.5.4.1, migratory caribou may be present on the site for as little as 1 week, and their short exposure to site-related CoPCs does not translate into adverse effects. Therefore, the uncertainty associated with the time-use estimate used in the exposure model represents an overestimation of site exposure for the majority of caribou in the WACH. However, the time use estimate of 5 months is appropriate for the very small proportion of the herd that potentially over-winter at the site.

The potential for elevated concentrations to occur during the period of snowmelt has been preliminarily assessed in a USGS study by Brabets (2004). The study found no exceedances of drinking water or aquatic life standards in stream water or snow samples. Therefore, wildlife that utilize the DMTS during periods of snowmelt would not likely be acutely affected through dietary exposure. Nevertheless, the possible need for future studies will be evaluated during development of the risk management plan, as described in Section 8.3.

6.6.5.1.4 Area Use

Area use for wildlife receptors was addressed in the risk assessment by modeling exposures according to the size of their home ranges (Table 6-30). Point estimates of exposure for small mammals, songbirds, and stream and pond receptors were approximations of the average exposures individuals would receive if their home ranges were centered at a sampling station. These estimates are uncertain, because they were generally based on point concentrations that were assumed to be representative of the actual exposure a receptor would receive throughout its home range, and may not capture the variability in CoPC concentrations in food or abiotic media across the home range. Exposure estimates for larger receptors like the caribou did account for the variability in tundra soil and tissue concentrations across their home ranges, because they were based on mean and 95%UCL on the mean concentrations for broad assessment units. However, whole site exposure scenarios for caribou and moose were biased toward higher exposures, because the density of tundra soil and moss sampling stations was higher in the port area and near the road, where CoPC concentrations are elevated, than the density of stations in the central portion of the road or at greater distances from the road (Figures 6-14 and 6-18). Additionally, the assumption that resident receptors such as the moose, snowy owl, and arctic fox derived all of their dietary exposure from the area around the DMTS is conservative, considering their large home ranges (Table 6-30) and the cyclic nature of prey abundance in the Arctic (snowy owl and arctic fox may travel great distances in search of food; for example, see Parmelee 1992, Chesemore 1975, and Eberhardt and Hanson 1978). Also, the exposure models conservatively assumed that all of the DMTS road corridor and port area provided productive habitat for wildlife receptors, even though areas near the road and port facilities, which tended to have higher metals concentrations, are disturbed by transportation activities, which may reduce their attractiveness to wildlife. However, there is some uncertainty as to the possibility of wildlife usage in near-road areas that could occur as a result of early snowmelt (Auerbach et al. 1997).

6.6.5.1.5 Measured CoPC Concentrations in Environmental Media and Prey

All CoPC concentrations used to estimate wildlife exposures were measured values, which avoided the uncertainties associated with highly conservative soil (or sediment) to biota transfer factors one might use to model CoPC concentrations. However, data were not always available for all CoPCs in all media at a sampling station, transect, or assessment unit (for example, as a result of the inability to capture appropriate prey species at a specific location), and uncertainty was introduced into the exposure estimate when data from other locations were applied in those food-web model calculations, or in a few cases, when metals concentrations in one tissue were substituted for concentrations in another related tissue in the model (see Appendix K). Also, exposure to CoPCs in drinking water was typically estimated for terrestrial receptors using water concentrations from the nearest tundra pond, which may have underestimated the actual CoPC concentrations in sources of drinking water near the DMTS road. Receptors such as small mammals, for instance, are probably drinking from pooled rainwater rather than from distant ponds. However, since water ingestion is such a minor component of total dietary exposure, the uncertainty associated with water data is not likely to affect risk conclusions.

There is also some uncertainty surrounding undetected results used to calculate exposures; a reported undetected value indicates that the true concentration of the analyte is somewhere between 0 and the limit of detection. In the risk model, all analyses with results reported as undetected were represented as one-half the detection limit, which may have underestimated or overestimated true concentrations, but by selecting a measure of central tendency, this is not likely to greatly bias results in one direction or the other.

All plant samples were unwashed prior to analysis, and measured CoPC concentrations reflect both the internal tissue concentrations and concentrations in adhered soil or sediment particles. Therefore, the inclusion of incidental soil or sediment ingestion as separate pathways in these exposure models may have resulted in an overly conservative estimate of exposure to these media for herbivorous wildlife.

Exponent (2005a) and Brabets (2004) both sampled sediments from the Omikviorok River and Aufeis Creek at the haul road. On average, the sediment concentrations for cadmium, lead, and zinc reported by Brabets (2004) are about twice as great as those reported by Exponent (2005a). Stream sediment samples collected by Brabets (2004) were sieved prior to analysis using a 0.063 mm screen, and were thus enriched relative to sediment samples collected as part of the ERA by Exponent (2005a). As such, the two sets of samples are not directly comparable, and the Brabets sampling methodology is not appropriate for use in the risk assessment. Since the time of the Brabets (2004) sampling events, Teck Cominco has completed survey, sampling, cleanup (where needed), and closure of former concentrate spill sites (Teck Cominco 2003e, 2005a), including those near Deadman and New Heart creeks. Although sediments and invertebrate communities were sampled and evaluated in five representative creeks along the DMTS road as part of the ERA data collection, Deadman and New Heart creeks were not among those sampled. Future monitoring needs (including the possible need for monitoring in these creeks) will be evaluated during development of the risk management plan.

Xanthates were not measured in the ore concentrate; however, the uncertainty associated with xanthates is limited. Xanthates are typically water soluble with short environmental half-lives, ranging from 2.5 to 4 days (Xu et al. 1988), which reduces long-term environmental concerns.

Xanthates have potential to induce toxicity by complexing with heavy metals, such as cadmium and lead, thereby resulting in increased concentrations of metals in organs and tissues in organisms (Boening 1998). While it is possible that residual xanthates (or other factors such as humic acids) may increase the availability of metals for uptake by plant or animal receptors, any such effect would be reflected in the media concentration data that were collected for the risk assessment. Therefore, the potential for xanthates to potentially affect any receptor groups would already be inherently accounted for in the exposure calculations. In addition, the bioavailability of all metals except lead was assumed to be 100 percent in the ERA and HHRA. For lead, risk estimates were made using site-specific bioavailability information as well as EPA default values. The site-specific lead bioavailability information was developed from a 1993 NTP study using Red Dog lead concentrate (Arnold and Middaugh 2001). Bioavailability data from this study would have incorporated any effect on bioavailability of residual xanthates in the concentrate.

6.6.5.1.6 CoPC Bioavailability

Gastrointestinal absorption was assumed to be 100 percent in the risk estimation. However, all of the CoPCs identified in the screening-level risk assessment are elemental, are natural constituents of the soil matrix, and would have limited but varying degrees of absorption if ingested by a receptor. Results from recent bioaccessibility testing work (Shock et al. 2007) suggest that bioavailability of aluminum and barium in tundra soil at the mine area would be on the order of 4 percent and 19 percent, respectively, but in the risk assessment, bioavailability of metals in soils was assumed to be 100 percent. In addition, the absolute bioavailability of lead in Red Dog ore is only about 6.8 to 13.5 percent (Arnold and Middaugh 2001). Chemical solubility is an important factor in absorption efficiency, as is potency. In the absence of site-specific data on bioavailability, the risk model assumed that the form of the chemical present in the environment was absorbed with the same efficiency as the chemical form used in the laboratory study from which the TRV was derived. In addition, TRV studies for some apparent risk-driving chemicals, such as aluminum and barium in mammals, employed a drinking water exposure pathway, while receptors in the DMTS road corridor and reference areas received the majority of their dietary exposures to CoPCs through the ingestion of food and soil or sediment (see Appendix K), where chemicals are bound up in matrices and less available than dissolved species. Therefore, the assumption that both the environmental and tested forms of a chemical are absorbed with the same efficiency has resulted in a general overestimation of exposure and risk across the assessed receptors. Because these TRVs were used to evaluate both site and reference exposures, inflation of the risk estimates was somewhat controlled through the comparison of site and reference exposure scenarios, although hazard quotients elevated above reference results do not necessarily indicate unacceptable risk, particularly when the TRVs used to estimate risk were conservative, and the gastrointestinal absorption efficiency was assumed to be 100 percent. Uncertainties associated with individual TRVs are discussed in Section 6.6.5.4.

In the case of the ptarmigan, risk results suggest that adverse effects from barium and lead exposures may occur in herbivorous birds foraging near the mine, and that adverse effects from lead exposures are also possible near the port, particularly for the most exposed individuals in the population of birds at the port. In the central portion of the road, the likelihood of adverse effects to foraging herbivorous birds is low, as the 95%UCL on the mean exposures did not exceed LOAEL TRVs, and only exposure to barium exceeded the NOAEL TRV (hazard quotient of 1.7). In the case of lead, however, more than 90 percent of the exposure is attributable to lead in soil. As mentioned above, the food-web models assumed 100 percent bioavailability of metals. However, site-specific bioavailability studies using rats have shown the bioavailability of lead in Red Dog ore to be only about 20 percent that of the soluble lead used in the studies on which the TRV is based (ADPH 2001; Arnold and Middaugh 2001; Arnold et al. 2003). If the relative bioavailability of lead in tundra soil to ptarmigan is also about 20 percent, then all LOAEL-based hazard quotients for ptarmigan would be less than 1.0, even using the 95%UCL on the mean CoPC concentrations. Similar results might be expected for barium, if site-specific bioavailability values were available for use in the food-web models.

Although there is some uncertainty involved in extrapolating results across taxonomic classes, these rat results suggest that the food-web models substantially overestimate lead bioavailability to the ptarmigan. This assumption is based on the fact that lead bioavailability is dependent on acid dissolution in the gut, which can be controlled by pH of the stomach and residence time of food in the stomach. Birds must be as efficient as possible at ingesting and digesting food, and therefore the digestive system of birds has adaptations designed to facilitate flight, such as a shorter intestinal tract in birds relative to mammals (Denbow 2000). Birds also typically have lower retention times in the gastrointestinal tract than mammals (Stevens and Hume 1995). For example, Stevens and Hume (1995) report mean fluid and particle retention time for a rock ptarmigan at 9.9 and 1.9 hours, respectively. In contrast, the rat has a much longer fluid and particle retention time of 20 and 22 hours, respectively (Stevens and Hume 1995). Therefore the longer retention time associated with the rat stomach would suggest higher relative bioavailability of lead in soil to the rat. In addition, acid secretion of birds is nearly equivalent to the rat, and more specifically, the pH of gastric juice in the ptarmigan (pH = 2.6, McLelland 1979) is nearly equivalent to that of the rat (pH = 2.7, Chu et al. 1999). Essentially equivalent pH but a much lower residence time of food or soil in the gastrointestinal system of the bird stomach compared to a mammal suggests that the relative bioavailability of lead would be lower for a bird. Therefore, the suggestion above that bioavailability of lead in tundra soil to ptarmigan is about 20 percent, similar to bioavailability for the rat (as mentioned above), is a reasonable and conservative approach to extrapolating results from the rat to the ptarmigan.

6.6.5.2 Reference Exposure Estimates

The uncertainties surrounding chemical concentrations and exposure parameters described in the previous section also apply to exposure estimates in the terrestrial reference area and reference lagoons. Reference exposures were calculated in order to quantify background risk levels to compare with site levels and to represent offsite exposure in risk calculations for migratory receptors. Multiple stations were sampled in the reference areas in order to capture the natural variability in metals concentrations, but there is some uncertainty in reference exposure estimates because of relatively small sample sizes (Appendices C and G). As estimates of offsite exposures for migratory receptors, reference exposures may overestimate or underestimate risk, depending on the metals concentrations encountered in the actual offsite habitats.

6.6.5.3 Representativeness of Sampling Locations

Prey or food tissue samples were collected at multiple locations along the length of the DMTS road in order to evaluate spatial variation in risk to receptors. Prior sampling of environmental media (i.e., tundra soil and road soil) had indicated that CoPC concentrations were generally higher at each end of the road, near dust sources at the port facilities and mine, and lower in the central portion of the DMTS road. Sampling had also indicated that CoPC concentrations tended to decline with increasing distance from the road. Tissue sampling locations were established to span these two-dimensional gradients. Results indicate that in general, CoPC concentrations in tissue samples reflected this gradient (see Figures 4-10 through 4-12, and discussion in Section 4.2.1). Some limited exceptions were noted, such as the one small mammal collected in the middle of the DMTS road with an anomalous mercury concentration. However, overall, tissue data from the locations sampled were adequate to detect spatial patterns in the magnitude of risk to wildlife receptors, including, in general, return to background levels of risk. Therefore, locations sampled appear to be representative of the predicted pattern of exposure along the DMTS road, and their number and layout appears to be sufficient for risk assessment purposes.

6.6.5.4 Toxicity Reference Values

Availability of toxicity data and suitability for use at a given site vary on a case-by-case basis. The selection of TRVs used in this assessment was based on an evaluation of the technical quality and ecological relevance of the study from which the values were taken. Modeled exposures were compared directly with the best available NOAEL and LOAEL TRVs derived from the literature, as outlined in the effects characterization (Section 6.5.2). The best available TRVs were selected based primarily on dietary exposure studies rather than drinking water exposure studies. Dietary exposure studies were preferred because drinking water ingestion was a very minor exposure route for wildlife receptors in the vicinity of the DMTS road corridor and reference areas. Those receptors receive the majority of their dietary exposures to CoPCs through the ingestion of food and soil or sediment (see Appendix K).

Nonetheless, some TRVs for apparent risk-driving chemicals are overly conservative, because they are based on exposures to chemicals dissolved in drinking water, which are highly available, and as discussed above (Section 6.6.5.1.6), drinking water ingestion was a very minor exposure route for wildlife receptors. Dietary exposure to these CoPCs at the site would likely result in lower toxicity than a comparable exposure rate to dissolved forms in drinking water. In particular, aluminum and barium TRVs for mammals resulted in high hazard quotients disproportionate to the likelihood of adverse ecological effects. The mammalian NOAEL and LOAEL TRVs for aluminum were based on significant reductions in weight gain of second- and third-generation mice exposed to aluminum chloride dissolved in drinking water (Ondreicka et al. 1966; see Section 6.5.2), and because the experiment tested only one dose, an uncertainty factor of 0.1 was applied to the LOAEL to predict the NOAEL TRV. Thus, the magnitude of ecological effects to mammalian receptors is likely to be lower than expected based on the TRV study, especially considering that this TRV also predicts adverse effects to mammalian receptors at background aluminum concentrations. Additionally, the aluminum in the environment at Red Dog is predominantly in the form of mica and feldspar (Clark 2005, pers. comm.), which contain aluminum silicates, a relatively insoluble form of the element. The Agency for Toxic

Substances and Disease Registry toxicological profile for aluminum notes that absorption of less soluble, less bioavailable forms of aluminum can be up to 10-fold lower than absorption for soluble forms, such as chloride (ATSDR 1999c). This factor is another reason why the TRV used in this assessment probably represents an overly-conservative estimate of the toxicity of aluminum in the environment.

The barium TRVs used in risk calculations for mammals were also derived from studies of chemical exposure in drinking water (dissolved barium chloride), which, like the aluminum studies, tended to overestimate the dietary assimilation of the CoPC in the field. Additionally, in a toxicological profile of barium, ATSDR (1992b) note that when evaluating the health effects of barium, differential solubility of barium compounds is important. For example, in soluble forms of barium (such as chloride, nitrate, and hydroxide) the Ba^{2+} ion is highly available and these compounds are highly toxic to humans and test animals. However, insoluble compounds (such as sulfate and carbonate) are generally non-toxic because the Ba^{2+} ion is unavailable. Barium in the concentrate at Red Dog is mostly in the form of barium sulfate, with a small amount of barium carbonate (Clark 2005, pers. comm.). Therefore, this indicates that the barium TRV, derived from a highly soluble form of the element, probably greatly overestimates the toxicity of the form in the environment at the study site. Recently, EPA released an Interim Final Ecological Soil Screening Level document for barium that recommended a mammalian NOAEL TRV of 51.8 mg/kg-day (U.S. EPA 2003a). This value represented the geometric mean of NOAEL results for growth and reproductive effects reported in studies reviewed by EPA and is an order of magnitude higher than the NOAEL TRV used in the risk assessment, perhaps reflecting the conservative nature of the latter study's exposure route and consideration of less toxic forms of the element. Food-web model results for tundra vole, caribou, tundra shrew, and muskrat showed incremental risk from barium exposure (Section 6.5), but if exposures to barium were compared against EPA's NOAEL TRV, all hazard quotients for caribou and muskrat and most hazard quotients for vole and shrew would drop below 1.0. Based on this comparison, all vole hazard quotients for barium would be less than 2, and all shrew hazard quotients would be less than 3. When assessed against an effects threshold determined through a weight of evidence of multiple studies as opposed to a single value, barium exposure estimates for mammals suggest that potential adverse effects are limited to small mammals very close to the road and near the mine.

Conservative assumptions were also made in the selection of chromium and mercury TRVs, which were based on exposures to chromium[VI] (for mammals) and methylmercury (for birds and mammals), respectively. These chemical species are more toxic than other forms of the elements that are prevalent in the environment, such as chromium[III] and inorganic mercury compounds (ATSDR 2000; USGS 2002). Speciation of chromium and mercury was not performed at the site, but these metals most likely occur in multiple forms, and therefore the selected TRVs are protective for the worst-case scenario and tend to overestimate risks.

As mentioned above, efforts were made to select the best available TRVs, based on appropriate exposure studies and most relevant endpoints. For example, if both drinking water and dietary exposure studies were available, the dietary exposure study was selected preferentially. U.S. EPA (2005) recommended a mammalian lead NOAEL TRV of 4.7 mg/kg-day. The mammalian NOAEL for lead recommended by U.S. EPA (2005) was based on a drinking water study, and was therefore not an appropriate TRV based on the selection criteria. Additionally, deriving

TRVs from exposure studies that are focused on chemicals dissolved in drinking water, which are highly available, is overly conservative and would overestimate exposure. For lead, a dietary exposure study was available, and therefore the mammalian NOAEL TRV used in this risk assessment was based on the more appropriate dietary study. Similarly, U.S. EPA (2005) recommended an avian lead NOAEL TRV of 1.63 mg/kg-day. The avian NOAEL for lead recommended by U.S. EPA (2005) was based on a paper that used Japanese quail as the receptor and the number of eggs produced as the relevant endpoint. Japanese quail have been bred specifically to have unnaturally high egg-laying rates, and therefore the relevance of "egg production" as the endpoint for wild birds is unclear. The meaning of extrapolating any apparent reproductive "effect threshold" in quail to wildlife receptors is unknown and highly questionable, because of differences in reproductive physiology. Therefore, a NOAEL TRV was selected that was derived from a study (Pattee 1984; see Section 6.5.2.9) that used wild species (American kestrels), dietary exposure, and the relevant endpoints included body weight, food consumption, clutch initiation, interval between eggs, clutch size, fertility and eggshell thickness. Thus, it should be noted that if the U.S. EPA (2005) NOAEL had been used, the NOAEL hazard quotients would have been higher.

A few CoPCs could not be eliminated at the screening stage because of a lack of appropriate TRVs; these chemicals include iron and silver for all wildlife receptors, and antimony, cobalt, and strontium for birds (summarized in Section 3.6.3). In the absence of toxicity measures, risks to birds and mammals from exposures to iron and silver, and risks to birds from exposures to antimony, cobalt, and strontium, were not evaluated quantitatively in the baseline risk assessment; therefore, there is some uncertainty about their potential to cause adverse ecological effects. Statistical comparisons of metals concentrations in site and reference media showed that iron was not significantly higher in tundra soil, tundra pond sediment, or lagoon sediment at the site than at reference locations, but that iron was elevated in site stream sediment relative to reference stream sediment (Tables 3-4, 3-6, 3-8, and 3-10). Silver concentrations were significantly higher than reference levels in site tundra soil and stream sediment but not in coastal lagoon sediment. The comparisons indicate a higher potential for iron or silver exposure in some environments at the site than in comparable reference areas. Thus, incremental exposure to these chemicals cannot be discounted, but the toxicological significance to exposed wildlife is not known.

Although risks to birds from exposure to antimony, cobalt, and strontium were not assessed quantitatively, screening-level food-web models for tundra vole eliminated strontium from further evaluation but retained antimony and cobalt as CoPCs for terrestrial wildlife (Section 3.5.6.1; Table 3-30). Risks to mammals from antimony and cobalt exposures were evaluated in the toxicity assessment (Section 6.5.3); no mammalian hazard quotients exceeded 1.0 for antimony, and only one NOAEL-based TRV exceedance was observed for cobalt (muskrat in tundra pond TT1-1000). Risk results for mammals indicate that these metals are very unlikely to cause adverse effects in mammalian receptor populations. Unless birds have much greater sensitivity to antimony, cobalt, or strontium, mammalian results suggest that adverse effects to avian populations are also unlikely.

The modeling technique used in the risk assessment evaluates each chemical individually, because the TRVs used for evaluating the ecological significance of exposure are also chemical-specific. Chemical-specific hazard quotients calculated by this method permit identification of specific chemicals that may cause adverse effects in ecological receptors. Simultaneous exposure to multiple chemicals could produce cumulative effects that are greater than the effects predicted for individual chemicals. Simple approaches such as summation of individual hazard quotients to calculate a hazard index are sometimes used to estimate cumulative effects; however, this assumes effects are additive, which may not be true based on the chemical-specific modes of action, and may be an overly-conservative approach if some metals act antagonistically. Although it is possible that interactions between combinations of metals could result in differences in bioavailability and/or toxicity relative to individual metal exposures, these potential interactions have been poorly characterized in the literature, at best. Furthermore, the effect of the interaction could be positive or negative. For example, zinc can reverse cadmium-induced toxicity (Peraza et al. 1998). Without a thorough understanding of the mechanisms by which individual metals elicit toxicity and the synergistic and antagonistic interactions between those metals (e.g., mode of action and target organ for each chemical in each receptor), a simple summation of hazard quotients could either underestimate or overestimate additive effects, which would convey little useful information beyond that presented in the individual chemical hazard quotient results.

6.6.5.5 Uncertainty in TRV Extrapolation

The range of toxicity thresholds reported in the literature for different test species is very large, even among those studies deemed suitable for extrapolation to the receptor species of interest. Consequently, uncertainty exists for extrapolated TRVs.

Observational errors in conducting toxicological experiments from which a TRV is derived stem primarily from parameter uncertainty. Uncertainty in TRV extrapolation, which may arise because of suspected differences in physiological responses of organisms to chemical exposures under identical conditions, is the result of model uncertainty.

Until recently, EPA has not used quantitative evaluation of uncertainty. Instead, conservative generalizations have been substituted in risk assessments to account for uncertainty and variability. An example is the use of generic “uncertainty factors,” which are intended to estimate the lower bound of the entire spectrum of possible toxicity levels. Although easy to use, such an approach is highly subjective. Uncertainty factors provide no information with regard to the character of risk. They are assumed to introduce a level of conservatism into the analysis, but provide no basis for verification. Therefore, uncertainty factors may just as easily underprotect as overprotect a defined receptor population. Furthermore, given their arbitrary derivation and application, they do not improve the accuracy or precision of the TRV.

The deterministic hazard quotient method is precise and justifiable based on observational investigations. Analysis of the available literature provided no reason to assume that the receptors evaluated in this investigation would be more sensitive to CoPCs than those tested in the respective toxicity studies cited. Some risk assessors have used the concept of allometric scaling (i.e., scaling by body weight) to adjust TRVs based on laboratory test species to what are perceived to be more appropriate values for wildlife. The conceptual basis for allometric scaling of TRVs is that the rates of many physiological functions, including toxicokinetics and toxicodynamics, vary across species in a non-linear fashion with body surface area (where body weight is used as a surrogate when surface area is unknown). The use of allometric scaling is controversial. Opponents of the method argue that chemical toxicity is complex and not necessarily predictable on the basis of the relative rates of metabolic processes (ept 1996). 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, there is no strong evidence for application of scaling factors other than 1 for chronic avian or mammalian TRVs for metals.

Allometric scaling most accentuates putative differences in sensitivity when the body mass of a wildlife species is much heavier (or lighter) than the body mass of the test species, such as the difference between a moose, as the heaviest mammalian receptor assessed, and a mouse or a rat that frequently is the test species in studies that are used for TRV derivation. Sample and Arenal (1999) recommend a scaling factor of 0.94 for mammals.

Because of the nature of the allometric scaling equation, application of this factor produces lower TRVs for heavier mammals (Table 6-35). For example, the LOAEL for lead of 90 mg/kg-day for rats corresponds to a LOAEL of 60 mg/kg-day when scaled to a moose’s body weight. To determine if scaling would produce different conclusions regarding risk, exposure estimates for moose were compared with allometrically scaled TRVs. Using non-scaled TRVs, results for moose indicated exceedances of NOAEL hazard quotients for aluminum at all sites, and for barium at the road site, but there were no LOAEL hazard quotient exceedances for any of the analytes at any of the sites. Using scaled TRVs, there were no additional exceedances of NOAEL or LOAEL TRVs, with the exception of barium at the mine assessment unit, but only based on the 95%UCL on the mean exposure scenarios; the mean exposure scenario did not indicate exceedances of barium using scaled TRVs. There are no exceedances of NOAEL or LOAEL hazard quotients for cadmium, lead, and zinc using either scaled or non-scaled TRVs.

To further explore the nature of the allometric scaling equation for mammalian TRVs, shrews were also examined, because shrews are lighter than the test species, as opposed to the moose (above) which is heavier than the test species. Scaled TRVs increased for the shrew receptor, resulting in decreases in all hazard quotients. The greatest difference for shrews is that for a number of stations where NOAEL-based hazard quotients slightly exceeded 1.0 based on non-scaled TRVs, the corresponding hazard quotients are less than 1.0 when scaling is applied. Specifically, these changes occur for arsenic (at TT5-0010, TT5-0100, TT2-0010, TT3-0010, and TT6-0010), mercury (at TT2-0100 and TT2-1000), selenium (at TS-REF-5 and TT2-0100), and vanadium (at TT2-0100). Also, there were no exceedances for NOAEL-based hazard quotients when scaled TRVs were used for cadmium (at TT5-2000 and TT2-0100) and zinc (at TT5-0100, TT5-2000, and TT2-0010), and there were no changes for lead. In addition, LOAEL-based hazard quotients are less than 1.0 for barium (at TT2-0100 and TT3-0100) and selenium (at TT5-0010) if allometric scaling is applied. Therefore, when scaled TRVs are used to determine hazard quotients for the moose and shrew, which are the heaviest and lightest mammalian receptors examined in this risk assessment, the range of results suggests that scaled

TRVs would indicate decreased risk for the shrew, and no changes in risk estimates for the moose, with the exception of barium at one site, and this exception occurs only when using the 95%UCL on the mean concentration.

For birds, the application of the scaling factor (1.2) recommended by Sample and Arenal (1999) produced the opposite trend (Table 6-36). For birds, TRVs increased for birds that are heavier than test species, but decreased for lighter wild species. Lapland longspurs, which weigh less than test species, had slightly higher hazard quotients using allometrically scaled TRVs. The greatest difference based on scaled TRVs is that NOAEL-based hazard quotients equal or slightly exceed 1.0 for mercury and zinc at all stations, including the reference area. However, the ranges of the hazard quotients at site stations (0.98–1.9 for mercury and 1.3–2.3 for zinc) are comparable to hazard quotients at the reference station (1.2 for mercury and 1.4 for zinc), indicating that incremental risk is negligible. There were no exceedances of cadmium, lead, and zinc results for Lapland longspurs using both scaled and non-scaled TRVs. Using scaled TRVs, all hazard quotients would decrease slightly for snowy owls, which are heavier than test species, but these changes have no significant effect on risk estimates for any of the analytes, including cadmium, lead and zinc.

Overall, results indicate that conclusions regarding risk to wildlife species are largely unchanged whether or not allometric scaling is applied to TRVs, and when scaled TRVs are used, results typically change in the direction of less risk, although scaling increases risk estimates for some receptors and lowers estimates for others. Therefore, not applying scaling factors does not bias or increase uncertainty in risk estimates for receptors.

6.6.5.6 Population Level Uncertainty

Toxicity bioassays, such as those conducted on laboratory test species in studies from which TRVs are derived, measure direct effects on individuals. As such, they give an indication of the physiological response of individuals to the tested conditions. Toxicity bioassays cannot account for any ecological factors or responses to chemical challenges by a receptor population, which may alter the ultimate effect at the population level. Such factors, which may potentially moderate or accentuate effects predicted by bioassays, include acclimation, feedback control of local receptor populations through adjustment of vital rates (e.g., birth, immigration, emigration), and inter-species interaction and dependency.

The implicit assumption in the assessment is based on the responses of individuals. The hazard quotient approach presumes that an exposure level associated with individual effects is absolutely consistent (i.e., lacking in natural variability) and is likely to cause demographic effects on a wild population. Although there is uncertainty associated with these assumptions, the conservative nature of the selection of input parameters for individual exposure scenarios should result in a conservative risk assessment when considering population-level effects.

An additional uncertainty related to estimating the potential for population-level effects relates to the appropriate definition of what constitutes a population for the receptors being evaluated. For example, as noted above, caribou present at the site, either as migrants or winter residents, are part of a herd (the WACH) that moves over vast areas of western Alaska. As discussed previously, it is inappropriate to extrapolate results of individual-based food-web models to conclude population-level effects without putting those results into context with regard to the proportion of the entire WACH population that is potentially exposed to CoPCs at the site. Similarly, although moose do not migrate like caribou, their home ranges can be large, as much as 5 to 10 km^2 (Wilson and Ruff 1999), and they can make seasonal movements up to almost 100 km during calving, rutting, or wintering (DFG 2003e). Therefore, creek- or lagoon-specific assessments, as were performed for moose, may be conservative with respect to risks to any individual moose, given their home range size in relation to the areas of lagoons and streams from which samples were collected, and even more conservative with respect to the larger moose population that frequents habitats within and beyond the DMTS assessment area.

Food-web model results for small-home-range receptors, such as shrews and voles, indicate the potential for adverse effects primarily within localized areas (e.g., within 100 m of the road, or around the mine boundary). These adverse effects to individuals, if occurring, could produce detectable higher-level responses, such as decreased population abundance or increased mortality, within these localized areas. However, the individuals in these localized areas are components of larger meta-populations. For example, it is very likely that voles move and disperse near as well as away from the road. Therefore, effects to individuals near the road would probably only translate into population level effects over larger areas (e.g., square kilometers of tundra) if habitats near the road represented a population “sink” where local environmental factors, including CoPCs, did not permit reproduction to occur at the replacement rate, and immigration of migrants from other sub-populations resulted in an overall decrease in abundance at the meta-population level. No population data are available confirm or deny the existence of such a sink near the road or mine. Therefore, there is considerable uncertainty that putative effects to individual small mammals living in habitats near these features would produce detectable population-level changes over broader spatial scales (within a kilometer from the road, within the CAKR, etc.). Broad-scale population surveys would be required to determine whether impacts to populations are occurring over these larger spatial scales.

6.6.5.7 Uncertainty in Risk Characterization

The method of risk characterization used in this assessment involved the direct comparison of estimated exposures at various locations across the site with reported TRVs. This is a snapshot approach to risk estimation. The determinant analysis applied here was structured to represent a threshold protective of the average individual in the population, and where possible, the proportions of the population represented by the 95%UCL on the mean CoPC concentration. The latter is a worst-case estimate based on the variance observed between sampling stations, while the former is more ecologically realistic but less protective of more vulnerable individuals (those with the highest exposures). Although risk scenarios were developed for multiple receptors in multiple areas along chemical gradients, with high CoPC concentrations near the DMTS road, port, and mine operation, and lower concentrations up to 2 km from sources (Appendices C and G), and although the scenarios spanned the extent of the DMTS road corridor, a great deal of extrapolation or extensive additional data collection would be required to translate deterministic hazard quotient results into quantitative estimates of the proportion of the exposed receptor populations that could be adversely affected.

6.7 Interpretation of Ecological Significance

In the previous sections, risk has been evaluated for ecological receptors inhabiting terrestrial, freshwater, and coastal lagoon ecosystems. Risks have been characterized based on empirical data collected during field investigations, food-web models, and comparison to findings reported in the scientific literature. For each receptor, the major uncertainties related to exposure or effects assessments have been identified, and their impacts on risk results have been assessed. This section compiles the risk characterizations for individual receptors to address the ecological significance of fugitive dust releases to the ecological communities studied in this ERA: terrestrial habitats, freshwater stream and pond habitats, and coastal lagoon habitats. Summaries of results discussed below are presented in Tables 6-37 through 6-44.

6.7.1 Terrestrial Habitats

Effects are observable on coastal plain and tundra plant community structure within 100 m of the DMTS road, primarily including reduced evergreen shrub, moss, and lichen cover (Tables 6-37 through 6-40). However, at 1,000 m from the road, communities were generally similar to reference communities except for a 2 to 4.5–fold difference in lichen cover. Lichen covers at stations TT5-1000 and TT5-2000 near the port were 2.75 and 8.25 percent, respectively, as compared to 15.75 percent at the coastal plain reference station, and lichen covers at stations TT3-1000 and TT8-1000 along the road were 4.75 and 5 percent, respectively, as compared to 9.75 and 21.8 percent at comparable reference stations. Community shifts within the first 100 m appear to be partly a result of physical influences of the road and their effect on hydrology, soil chemistry, and plant vitality. Deposition of CoPCs in fugitive dust probably also contributes to observed changes in community parameters within the first 100 m, which are interrelated with, and similar to, the effects caused by physical and chemical stressors common to other gravel roads in tundra environments. Beyond 100 m, differences observed between reference and site communities, specifically the decrease in lichen cover, appear to be a result of dust deposition, as non-vascular plants are apparently more sensitive to road dust and to metals in dust than vascular species. The lichen cover values at 1,000-m and 2,000-m stations, which were significantly lower than reference cover values, indicate that lichen effects are present at these distances from the DMTS road corridor, and perhaps beyond. The contribution of metals in producing some of these effects, particularly on plant communities near the DMTS road, is unclear. Further study would be required to define the full nature and extent of lichen effects beyond 1,000 to 2,000 m and to distinguish the relative contributions of causative agents, such as metals and road dust or other factors on lichen toxicity. In port facility areas, particularly in the area immediately downwind of CSB1, the presence of stressed and dead vegetation appears to be primarily related to fugitive concentrate dust deposition.

Hazard quotient results for wildlife receptors are summarized in Tables 6-41 through 6-43. Locations and receptors where NOAEL and LOAEL hazard quotients, or only LOAEL hazard quotients exceeded 1.0 are summarized in Tables 6-41 and 6-42, respectively. Table 6-43 summarizes the number of LOAEL hazard quotient exceedances per number of sites evaluated for each receptor, providing a broad overview of the results. A discussion of results follows.

Herbivorous and insectivorous small mammals (i.e., tundra vole and tundra shrew) inhabiting tundra within 10–100 m of the DMTS road, near the port facility areas or near the mine's ambient air/solid waste boundary showed incremental risk from exposure to aluminum and barium. By 1,000 m, hazard quotients were generally below 1.0 and/or comparable to reference area hazard quotients. No other CoPCs had LOAEL-based hazard quotients greater than 1.0 for these receptors (Table 6-43). Therefore, if adverse effects occur for small mammals, they are most likely to exist in localized areas near facilities or within a narrow band of tundra about 100-m wide near the road as a result of exposure to aluminum and barium. However, despite the elevated hazard quotients for these two metals, adverse effects are considered to be unlikely, given the conservative nature of the aluminum and barium TRVs and low bioavailability of aluminum and barium at the site (Shock et al. 2007).

The food-web model results for terrestrial herbivorous birds (i.e., ptarmigan) suggest that adverse effects (mortality or reproductive effects) from lead exposure near the port and barium and lead exposure near the mine are possible. These effects, if occurring, could result in population-level effects in areas near the port or mine. Along the length of the road, the likelihood of adverse effects to herbivorous birds foraging in these areas is low, as 95%UCL on the mean exposures did not exceed NOAEL or LOAEL TRVs (Table 6-42), except for exposure to barium, which exceeded the NOAEL TRV (hazard quotient of 1.7). However, considering the conservative barium TRV and bioavailability assumptions, and the low bioavailability of barium at the site (Shock et al. 2007), barium is unlikely to pose a risk to ptarmigan in any of these site areas.

For over-wintering caribou evaluated in the food-web models, LOAEL-based hazard quotients for aluminum ranged from 2.2 to 2.5 across the site, about 3-fold higher than comparable reference area hazard quotients. The LOAEL-based hazard quotient for barium was 1.3 in the near-mine area. Based on the low proportion of the total herd that could possibly over-winter near the mine site, it is very unlikely that any individual-level effects (e.g., reduced growth) would lead to population-level effects for the entire WACH, most of which visit the site only briefly during migrations. Considering the conservative aluminum and barium TRV and bioavailability assumptions, and the low bioavailability of aluminum and barium at the site (Shock et al. 2007), no adverse effects are anticipated for individual over-wintering caribou.

Food-web model results (Tables 6-41 through 43) also indicate that exposure to CoPCs is unlikely to result in population-level effects to other large-bodied mammalian herbivores (e.g., moose), or to avian invertivores (e.g., Lapland longspur). Population-level effects are also unlikely for avian and mammalian carnivores (e.g., snowy owl and arctic fox), as discussed in the Risk Characterization (Section 6.5.4) and considering the conservative aluminum and barium TRV and bioavailability assumptions, and the low bioavailability of aluminum and barium at the site (Shock et al. 2007).

Overall, results of the ERA show that predicted adverse effects to wildlife receptors are largely restricted to localized areas adjacent to the DMTS road, the port facility, and the mine ambient air/solid waste boundary; however, effects on tundra vegetation apparently extend further, with effects on lichens observed at 1,000 to 2,000 m away from these dust sources and perhaps beyond. Table 6-44 summarizes observed and predicted ecological effects for receptors in each environment (terrestrial, freshwater, and coastal lagoon habitats) and in each area (near the port, along the road, and near the mine).

6.7.2 Freshwater Habitats

In general, adverse ecological effects are not predicted in streams that cross the DMTS road, based on multiple lines of evidence (Table 6-44). First, the evaluation of benthic macroinvertebrate drift assemblages indicated that the overall characteristics of the communities found in the three site stream stations were similar to reference streams. Second, fish monitoring studies have found relatively low metals concentrations in fish tissue from Aufeis Creek and Omikviorok River compared to streams near the mine, and no consistent evidence of a road effect on fish tissue metals concentrations in these streams (Ott and Morris 2004). Similarly, selenium concentrations in Anxiety Ridge Creek fish tissue were comparable at both upstream and downstream locations, while selenium concentrations were lower at the DMTS road station. In Anxiety Ridge Creek, where cadmium and lead concentrations in juvenile Dolly Varden tissue were significantly higher in downstream fish than upstream fish, maximum concentrations of cadmium and lead also exceeded the lowest literature thresholds for effects to survival, growth, or reproduction, but concentrations were also within the range of no-effects thresholds (Table 6-27). Therefore adverse effects to fish cannot be conclusively predicted, as the sensitivity of Dolly Varden relative to the test species is not known. Furthermore, maximum whole body fish tissue concentrations reported from a nearby naturally mineralized creek located north of the Red Dog Mine were higher or similar to concentrations reported for Anxiety Ridge Creek fish tissue. Third, metals concentrations in plants were within the range of reference concentrations (with the exception of aluminum and zinc in some willow leaf samples, and aluminum and chromium in sedges from the Omikviorok River) and in general, were not elevated in comparison to literature phytotoxicity thresholds. Fourth, food-web model results (Tables 6-41 through 6-43) indicated that exposure to CoPCs is unlikely to result in adverse effects to avian and mammalian herbivores (e.g., green-winged teal, muskrat, and moose) or avian invertivores (e.g., common snipe) foraging in the streams, as LOAEL-based hazard quotients were less than or equal to 1.0, or in the case of aluminum ranged from 1.8 to 8.3 for muskrat, but were comparable to reference area hazard quotients. The conservative aluminum and barium TRV and bioavailability assumptions used in the food-web models, and the low bioavailability of aluminum and barium at the site (Shock et al. 2007) should also be noted. Collectively, these findings indicate that no ecologically significant effects are likely in streams, with the possible exception of potential effects to fish in Anxiety Ridge Creek.

In general, adverse effects are not predicted in tundra ponds located greater than 100 m from the DMTS road and port facilities, with the exception of potential vegetation effects identified based on comparison to literature screening values at ponds situated in low-lying areas to the southwest of the mine's ambient air/solid waste permit boundary (Table 6-44). For ponds TP1-1000, TP3, and TP4, CoPC concentrations in sediment were less than the maximum no-effects concentrations for sediments from coastal lagoons that were evaluated in toxicity tests using freshwater test organisms. Vegetation around the ponds appeared to be healthy, and metals concentrations were within the range of reference concentrations (with a few exceptions for cobalt, lead, and zinc), and/or below phytotoxicity thresholds.

Incremental exposure to lead and zinc at pond TP4 (located along the road near the mine) resulted in minor exceedances of phytotoxicity thresholds in sedge tissue (Table 6-25). However, plant samples were not washed or rinsed prior to analysis. If they had been washed, concentrations may have been below effects thresholds. Also, the vegetation appeared healthy in observations made during field sampling. Given these considerations, adverse effects to vegetation are not expected in tundra pond TP4.

Tundra ponds observed at the site and reference area were hydrologically disconnected from surface water inputs from streams and are unlikely to support permanent fish populations. Therefore, pathways to fish and piscivorous wildlife are believed to be incomplete, and no adverse effects are expected for these receptors. Food-web model results indicate a very low likelihood of adverse effects to survival, growth, or reproduction of herbivorous wildlife potentially foraging at these ponds (Tables 6-41 through 6-43).

The possibility of adverse effects to invertebrates and plants could not be conclusively discounted at Station TP1-0100, located near the concentrate conveyor and other port facilities (Photograph 4). As described in Section 6.3.2, the likelihood of adverse effects to macroinvertebrates in TP1-0100 could not be evaluated, and phytotoxicity threshold comparisons for sedges showed a potential for vegetation effects from lead and zinc exposures. Aerial transport and surface flow are probably the main mechanisms by which metals in fugitive dust become deposited in this habitat, as is likely for the surrounding tundra. Ponds near the port facilities, such as TP1-0100, are not true ponds, but rather flooded depressions in the tundra, and may not be permanent as they are dependent on precipitation and surface runoff to maintain volume. The ephemeral nature of the port area ponds suggests that they would be less likely to support a diversity of ecological receptors than the larger, more permanent ponds that occur in the tundra along the DMTS road. Therefore, any adverse effects in these ponds have less ecological significance than if similar effects were to occur in ponds scattered across the tundra.

6.7.3 Coastal Lagoons

No adverse effects are predicted for ecological communities inhabiting coastal lagoons. Sediment toxicity tests indicated no effects to benthic invertebrates in lagoons, even when exposed to elevated CoPC concentrations in sediments from locations nearest to port facilities. Plant community structure was similar at site and reference lagoons and the few differences that were observed may reflect natural variability among and within lagoon plant communities, which fluctuate seasonally in size and composition as water levels rise and recede. However, plant community surveys were limited to the wetland vegetation at the perimeter of lagoons, and these results are not applicable to other coastal plant communities with different compositions, particularly those with abundant lichen cover. Plant communities with abundant lichen cover were assessed in the terrestrial coastal plain transects. Food-web model results (Tables 6-41 through 6-43) indicate that there is a very low likelihood of adverse effects on the survival, growth, and reproduction of herbivorous and invertivorous birds (e.g., brant and black-bellied plover) and herbivorous mammals (e.g., muskrat, moose) that potentially forage in the coastal lagoons. The lagoons evaluated in this risk assessment are not believed to support permanent fish populations because of their physical separation from potential marine and freshwater colonizing sources. Therefore, pathways to fish and piscivorous wildlife are believed to be incomplete, and no adverse effects are expected for these receptors. Collectively, these findings indicate that no ecologically significant effects are likely in coastal lagoons (Table 6-44).

7 Conclusions

The results of the risk assessment provide a snapshot of risk under current conditions that will help risk managers determine what additional actions may be necessary to reduce those risks now and in the future. The following subsections summarize the findings of the human health and ERAs. These sections are largely the same as the results sections in the *Executive Summary*, however, this section also includes reference to summary tables of results for the human health and ecological risk assessments.

7.1 Human Health Risk Assessment Conclusions

A site-specific HHRA (Section 5) was conducted to evaluate exposure to DMTS-related metals through incidental soil ingestion, water ingestion, and subsistence food consumption under three scenarios: 1) child subsistence use, 2) adult subsistence use, and 3) combined worker/ subsistence use. The estimated risks from each of the scenarios were within acceptable limits and are summarized below. Risks are necessarily expressed separately for lead and for the other (non-lead) metals because a different methodology is used to estimate lead exposure and risks, as described in Section 5.2.2.1. A summary of the lead modeling results was presented in Table 5-20, and summaries of the results for non-lead metals were provided in Tables 5-48 and 5-50. The findings are described below.

7.1.1 Child Subsistence Use

- Using the IEUBK model default soil lead bioavailability of 30 percent, the model predicted a geometric mean blood lead level of 1.2 μg/dL, with a less than 0.0005 percent chance of exceeding the target blood lead level of 10 μg/dL.
- Using the site-specific soil lead bioavailability of 9.7 percent, the model predicted a geometric mean blood lead level of 1.0 μg/dL, with a less than 0.0005 percent chance of exceeding the target blood lead level of 10 μg/dL.
- The cumulative hazard index from non-lead CoPCs was 0.3, well below the target hazard index of 1.0.
- Assuming a fractional intake from the site as high as 0.33 (which is 3.7 times the site fractional intake of 0.09), cumulative risks from non-lead CoPCs would not exceed the target hazard index of 1.0.
- The highest hazard index was 0.1 for cadmium exposure from caribou consumption. Assuming a fractional intake from the site as high as 0.95, caribou cadmium related risks would not exceed the target hazard index of 1.0.

- Assuming 100-percent intake from the site (fractional intake=1.0), no other single CoPC would have a risk exceeding the target hazard index of 1.0.

7.1.2 Adult Subsistence Use

- For subsistence use, lead risks were evaluated only for children, but this would also be protective of adult exposure (see results for lead summarized above for child subsistence use).
- The cumulative hazard index from non-lead CoPCs was 0.1, well below the target hazard index of 1.0.
- Assuming a fractional intake from the site as high as 0.93, cumulative risks from non-lead CoPCs would not exceed the target hazard index of 1.0.
- Assuming 100-percent intake from the site (fractional intake=1.0), no single CoPC would have a risk exceeding the target hazard index of 1.0.

7.1.3 Worker/Subsistence Use

- Using the ALM default soil lead bioavailability of 12 percent, the model predicted a geometric mean blood lead level in the fetuses of pregnant women of 1.9 μg/dL, with a 1.3 percent chance of exceeding the target blood lead level of 10 μg/dL.
- Using the site-specific soil lead bioavailability of 3.9 percent, the model predicted a geometric mean blood lead level in the fetuses of pregnant women of 1.6 μg/dL, with a 0.7 percent chance of exceeding the target blood lead level of 10 μg/dL.
- The cumulative hazard index from non-lead CoPCs was 0.08, well below the target hazard index of 1.0.
- Assuming 100-percent intake from the site (fractional intake=1.0), cumulative risks from non-lead CoPCs would not exceed the target hazard index of 1.0.

Overall, risks were well within public health acceptable limits. The results of the risk assessment, along with the results from the subsistence foods evaluations (Appendix H), suggest that risks associated with continued harvesting of subsistence foods from the site, including in unrestricted areas near the DMTS, are not significantly elevated. In addition, although harvesting remains off limits within the DMTS, human health risks were not elevated even when data from restricted areas were included in the risk estimates.

7.2 Ecological Risk Assessment Conclusions

A site-specific ecological risk assessment (Section 6) was conducted to evaluate risk to ecological receptors inhabiting terrestrial, freshwater stream and pond, coastal lagoon, and marine environments from exposure to DMTS-related metals. The risk conclusions for each habitat are summarized in the following sections and in Table 6-44.

7.2.1 Terrestrial Environments

- Changes in vegetation community structure are observable within 100 m of the DMTS road and port facilities. These community shifts appear to be, in part, a result of physical and chemical influences of the road and their effect on hydrology, soil chemistry, and plant vitality. Physical and chemical stresses are commonly found associated with gravel roads in tundra environments. The importance of CoPCs in fugitive dust relative to physical stresses caused by the DMTS road in producing these changes could not be determined based on the data available at this time. However, physical factors are likely to exert their greatest influence near the road and facility areas where dust deposition is greatest and drainage may be locally altered, whereas chemical factors (e.g., elevated metals and pH) are likely to become relatively more important at greater distances from dust sources, but may also be significant near the road and port facility areas.

- Differences between reference plant communities and plant communities beyond 100 m from the DMTS road, specifically the 2- to 4.5-fold decrease in lichen cover at 1,000 to 2,000 m from the road (Figure 6-4 and Tables 6-11 and 6-12), appear to be a result of fugitive dust deposition. Further study would be required to define the full nature and extent of lichen effects related to fugitive dust deposition from the DMTS port, road, and Red Dog Mine, and to identify the causative agent(s) of lichen decline.

- In port facility areas, particularly in the area immediately downwind of CSB1, the presence of stressed and dead vegetation appears to be primarily related to fugitive concentrate dust deposition.

- Herbivorous and insectivorous small mammals (e.g., voles and shrews) inhabiting tundra within 10–100 m of the DMTS road, near the port facilities, or near the mine's ambient air/solid waste boundary showed incremental risk from exposure to aluminum and barium. However, exposures decreased to no-effects levels or were comparable to reference exposures beyond 100 m from the road and 1,000 m from the mine's ambient air/solid waste boundary. Although elevated risks were predicted for aluminum and barium near the road, port, and mine, the actual potential for adverse effects is thought to be small given the highly conservative nature of the aluminum and barium TRVs and low bioavailability of aluminum and barium at the site (Shock et al. 2007).

- Adverse effects to herbivorous birds (e.g., ptarmigan) from lead are possible near the port and mine. These effects, if occurring, could result in population-level effects in these areas. However, along the length of the road, the likelihood of adverse effects to herbivorous birds is low.

- For caribou, no adverse effects are predicted for the vast majority (>99.98 percent) of caribou that pass through the site only during migration. Caribou over-wintering near the mine have an estimated exposure to aluminum and barium that is 1.3 to 2.5 times the LOAELs. However, the actual potential for adverse effects to over-wintering caribou is thought to be small, given the highly conservative nature of the aluminum and barium TRVs and low bioavailability of aluminum and barium at the site (Shock et al. 2007).

- Population-level effects are considered unlikely for other terrestrial wildlife, including large-bodied mammalian herbivores (e.g., moose), avian invertivores (e.g., Lapland longspur and common snipe), and avian and mammalian carnivores (e.g., snowy owl and arctic fox), under current conditions.

7.2.2 Freshwater Stream Environments

- Benthic macroinvertebrate drift assemblages indicated that the overall characteristics of the communities found in site streams crossing the road were similar to those in reference streams.

- Fish monitoring studies have found no evidence of a road-related effect on metals concentrations in tissue of fish upstream and downstream of the DMTS in the Omikviorok River and Aufeis Creek. However, in Anxiety Ridge Creek, near the mine, cadmium and lead concentrations in tissue of juvenile Dolly Varden were significantly higher in fish downstream from the haul road compared with upstream fish, and although the most conservative screening benchmarks for fish tissue were exceeded, concentrations were also within the range of no-effects values from the literature. Thus, adverse effects to fish populations are not predicted in the Omikviorok River and Aufeis Creek, but cannot be ruled out in Anxiety Ridge Creek.

- Metals concentrations in riparian area plants were generally within the range of reference concentrations and/or literature phytotoxicity thresholds. No indications of phytotoxicity were observed in plants at site streams, and plant health appeared similar at site and reference streams.

- The likelihood of adverse population-level effects to wildlife foraging in streams, including avian and mammalian herbivores (e.g., green-winged teal, muskrat, and moose) and avian invertivores (e.g., common snipe), is considered to be very low.

7.2.3 Freshwater Pond Environments

- Adverse effects are not predicted in tundra ponds along the DMTS road, or at distances greater than 100 m from facilities. For these ponds, CoPC concentrations in sediment are not expected to be toxic to benthic macrofauna based on toxicity test data for coastal lagoons. Metals concentrations in plants were generally within the range of reference concentrations and/or below phytotoxicity thresholds, and food-web models indicate a very low likelihood of adverse population-level effects to herbivorous wildlife (e.g., green-winged teal and muskrat) and avian invertivores (e.g., common snipe).

- There is a potential for adverse effects to invertebrates and plants in ephemeral ponds located within 100-m of the concentrate conveyor and other port facilities, although no effects were observed during field sampling in those ponds.

7.2.4 Coastal Lagoon Environments

- Sediment toxicity tests indicated no effects to benthic invertebrates in lagoons, even when exposed to elevated CoPC concentrations in sediments from locations nearest to port facilities.

- Plant community structure was similar at site and reference lagoons. Natural variability among and within lagoon plant communities likely accounts for the few differences that were observed. However, only fringing wetland vegetation was assessed for coastal lagoons, while plant communities with abundant lichen cover were assessed in the terrestrial coastal plain transects.

- The likelihood of adverse population-level effects to wildlife foraging in coastal lagoons, including herbivorous and invertivorous birds (e.g., brant and black-bellied plover), and mammalian herbivores (e.g., muskrat and moose), is considered to be very low.

- No fish were present in port site lagoons, as the lagoons have no open water connections to the Chukchi Sea, and they also freeze solid in the winter.

Marine Environment

- No effects were predicted for receptors in the marine environment because the metals concentrations in sediment and water were below effects levels.

8 Risk-Based Alternative Cleanup 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 and conditions are expected to change over time, it would be most practical to develop alternative cleanup levels following closure of Red Dog Mine, where appropriate. In the meantime, changes in conditions and in potential human and ecological exposures over the life of the operation can be addressed through implementation of risk management, control, and monitoring activities, as illustrated in Figure 1-1, which is the decision-making framework from DEC et al. (2002). A risk management plan will be developed to more clearly define the actions to be taken.

The approach described above will be protective of human health and the environment for the following reasons:

1. Human health risks were not found to be elevated (see Section 7, *Conclusions*). Nevertheless, conditions may change over time. The risk management plan will provide the means to monitor changes in conditions, and trigger additional actions, if needed, to control and minimize risks.
2. Ecological risks that were observed or predicted for some receptors (see Section 7, *Conclusions*) will be proactively addressed in the risk management plan. This plan will provide a variety of tools to monitor and minimize changes in conditions and pursue environmental improvements.

The approach to development of the risk management plan is described below.

8.1 Development of a Risk Management Plan

A risk management plan will be developed to address the issues identified by this risk assessment, which are summarized in Section 7, *Conclusions*. The plan will include evaluation of risk management options within the general categories of institutional controls, engineering controls, monitoring, and remediation/restoration. The plan will identify the most appropriate combination of actions to achieve the overall goal:

- **Overall Goal:** Minimize risk to human health and the environment surrounding the DMTS and outside the Red Dog Mine boundary over the life of the mine.[14]

[14] Note that the mine closure and reclamation plan will address risk management within the mine boundary.

In support of the overall goal, several preliminary objectives and action items are outlined below to address specific concerns identified in the ERA conclusions (Section 7.2). These include the following:

- Preliminary Objective 1: Reduce CoPC concentrations related to fugitive dust in nonvascular plants along the DMTS road and near the port and mine to decrease risk to these plants (particularly mosses and lichens), and to minimize exposure to wildlife that may consume these plants. Action items include the following:
  - Action 1: Tissue CoPC concentrations in mosses and/or lichens will be monitored to track the rate of change
  - Action 2: Moss and lichen community composition (e.g., diversity, abundance, cover, etc.) will be monitored at various distances from the DMTS road to track changes in moss and lichen condition in response to changes in fugitive dust deposition.
- Preliminary Objective 2: Reduce CoPC concentrations related to fugitive dust in shrubs and herbaceous plants to minimize exposure to herbivorous birds (and other wildlife) that may consume these food items in areas near the port and mine. Action item is as follows:
  - Action: Tissue CoPC concentrations in shrubs and/or herbaceous plants will be monitored to track the rate of change.

These objectives will be revisited and refined during development of the risk management plan. Additional objectives will be identified and defined as needed, and action items, including monitoring, will be further developed. Appropriate frequencies for monitoring will be defined. The frequency of monitoring could be increased or decreased in response to increases or decreases in the rate of change in CoPC concentrations. For example, monitoring frequency could be increased in response to increased mining activity (if an increased rate of change were observed); or monitoring frequency could be decreased in response to improved dust control (if a decreased rate of change were observed). In this way, increases or decreases in human and ecological exposures (relative to exposures evaluated in this risk assessment) can be closely monitored and managed through a decision process tied to these changes. To facilitate this evaluation in the future, Tables 8-1 and 8-2 provide a summary of concentrations in exposure media associated with the receptors that were found to be potentially at risk, i.e., vegetation (particularly more sensitive non-vascular plants such as lichen and moss), and herbivorous birds (i.e., ptarmigan). Table 8-1 provides a summary of mean CoPC concentrations in moss and lichen in port, road, and near-mine areas that could be used as a baseline with which to compare future monitoring data and assess changes in conditions. Table 8-2 summarizes mean lead concentrations in exposure media for ptarmigan, and includes the associated hazard quotient results in port, road, and near-mine areas.

Additional actions, such as use of engineering controls or remediation to reduce escape of metals-bearing dust into the environment, will also be identified during development of this plan to help achieve these objectives.

A variety of actions have already been taken to reduce risk of metals exposure from fugitive dust. For example, many measures have already been undertaken throughout mine, road, and port operations to reduce fugitive dust emissions, including significant improvements in engineering controls and operational procedures, as described in Section 2.2.4 (*Fugitive Dust Control Measures*). Soils containing elevated metals concentrations have been recovered and recycled to reduce the potential for exposure to occur, or for dust to be generated from these soils (Exponent 2002b). In addition, studies have been undertaken to evaluate areas of uncertainty, such as bioavailability (Shock et al. 2007) and weathering potential of metals in fugitive dust (Teck Cominco 2007b,c). Teck Cominco uses its environmental management systems program to define objectives and track progress for continuous improvement on their environmental performance, including with respect to fugitive dust emissions (e.g., see Teck Cominco 2007a). Current efforts in the mine area are summarized by Teck Cominco (2007d) and reported regularly at http://www.dec.state.ak.us/air/reddog.htm.

Development of the risk management plan is anticipated to be a collaborative process involving DEC and other stakeholders throughout the process of identifying, defining, and refining objectives, and evaluating options and methods to achieve those objectives.

9 References

ACAT. 2004. Analysis of reports on elevated levels of heavy metals in plants used for subsistence near Red Dog Mine, Alaska. Alaska Community Action on Toxics, Anchorage, AK.

ADPH. 2001. Public health evaluation of exposure of Kivalina and Noatak residents to heavy metals from Red Dog Mine. Alaska Division of Public Health, Department of Health and Social Services, Section of Epidemiology, and Environmental Public Health Program, Anchorage, AK.

ADPH. 2004. Public health evaluation and assessment biomonitoring of residents of Kivalina and Noatak lead and cadmium results. Department of Health & Social Services, Alaska Division of Public Health, Section of Epidemiology, and Environmental Public Health Program, Anchorage, AK.

ADPH. 2005. Evaluation and response to Alaska Community Action on Toxics' (ACAT) report: "Red Dog and Subsistence. Analysis of Reports on Elevated Levels of Heavy Metals in Plants Used for Subsistence near Red Dog Mine, Alaska." Department of Health & Social Services, Alaska Division of Public Health, Section of Epidemiology, and Environmental Public Health Program, Anchorage, AK.

AGRA. 2001. DeLong Mountain Terminal Project: Onshore facilities feasibility study. Draft Report. Volume 1. Project No. A151H. AGRA Engineering Global Solutions.

Aitchison, C.W. 1987. Review of winter trophic relations of soricine shrews. Mamm. Rev. 17(1):1–24.

Ambrose, A.M., P.S. Larson, J.F. Borzelleca, and G.R. Hennigar. 1976. Long-term toxicologic assessment of nickel in rats and dogs. J. Food Sci. Technol. 13:181–187.

Andreev, A.V. 1991. Winter adaptations in the willow ptarmigan. Arctic 44(2):106–114.

Anthony, R.M. 1997. Home ranges and movements of arctic fox (*Alopex lagopus*) in western Alaska. Arctic 50(2)147–157.

Anthony, R.M., N.L. Barten, and P.E. Seiser. 2000. Foods of Arctic foxes (*Alopex lagopus*) during winter and spring in western Alaska. J. Mammal. 81(3):820–828.

Arnold, S., and J.P. Middaugh. 2001. Bioavailability of Red Dog Mine lead ore. Assessment of Battelle study: Evaluation of lead ore in Fischer 344 rats, submitted to the National Toxicology Program, June 1993. Available at http://www.doerun.com/ENGLISH/html/leadinforeddog.htm.

Arnold, S.M., T.V. Lynn, and J.P Middaugh. 2003. Evaluation of exposure to heavy metals from Red Dog mine in northwest Alaska. Poster presentation at Society of Environmental Toxicology and Chemistry 24th annual meeting, Austin, TX. November 9–13, 2003.

ATSDR. 1992a. Toxicological profile for antimony. U.S. Public Health Service, Agency for Toxic Substances and Disease Registry, Atlanta, GA.

ATSDR. 1992b. Toxicological profile for barium and compounds. U.S. Public Health Service, Agency for Toxic Substances and Disease Registry, Atlanta, GA.

ATSDR. 1992c. Toxicological profile for thallium. U.S. Public Health Service, Agency for Toxic Substances and Disease Registry, Atlanta, GA.

ATSDR. 1999a. Toxicological profile for lead. U.S. Public Health Service, Agency for Toxic Substances and Disease Registry, Atlanta, GA.

ATSDR. 1999b. Toxicological profile for cadmium. U.S. Public Health Service, Agency for Toxic Substances and Disease Registry, Atlanta, GA.

ATSDR. 1999c. Toxicological profile for aluminum. U.S. Public Health Service, Agency for Toxic Substances and Disease Registry, Atlanta, GA.

ATSDR. 2000. Toxicological profile for chromium. U.S. Public Health Service, Agency for Toxic Substances and Disease Registry, Atlanta, GA.

Auerbach, N.A., M.D. Walker, and D.A. Walker. 1997. Effects of roadside disturbance on substrate and vegetation properties in Arctic tundra. Ecol. Appl. 7(1):218°235.

Aulerich, R.J., R.K. Ringer, M.R. Bleavins, and A. Napolitano. 1982. Effects of supplemental dietary copper on growth, reproductive performance and kit survival of standard dark mink and the acute toxicity of copper to mink. J. Animal Sci. 55(2):337–343.

Aulerich, R.J., A.C. Napolitano, S.J. Bursian, B.A. Olson, and J.R. Hochstein. 1987. Chronic toxicity of dietary fluorine in mink. J. Anim. Sci. 65:1759–1767.

Azar, A., H.J. Trochimowicz, and M.E. Maxfield. 1973. Review of lead studies in animals carried out at Haskell laboratory-two-year feeding study and response to hemorrhage study. pp. 199–210. In: Environmental Health Aspects of Lead. D. Barth, A. Berlin, R. Engel, et al. (eds). Commission of the European Communities, Luxembourg.

Barbour, M.G. J.H. Burk, and W.D. Pitts. 1980. Terrestrial plant ecology. The Benjamin/ Cummings Publishing Company, Inc., Menlo Park, CA. 604 pp.

Barbour, M.T., J. Gerritsen, B.D. Snyder, and J.B. Stribling. 1999. Rapid bioassessment protocols for use in streams and wadeable rivers: Periphyton, benthic macroinvertebrates and fish. Second Edition. EPA 841-B-99-002. U.S. Environmental Protection Agency, Office of Water, Washington, DC.

Bechtel Jacobs. 1998. Biota sediment accumulation factors for invertebrates: Review and recommendations for the Oak Ridge Reservation. Prepared for U.S. Department of Energy, Office of Environmental Management. Bechtel Jacobs Company LLC, Oak Ridge, TN.

Bee, J.W., and E.R. Hall. 1956. Mammals of northern Alaska. University of Kansas Publishing, Museum of Natural History. ~297 pp.

Beisel, J-N., P. Usseglio-Polatera, V. Bachmann, and J-C. Moreteau. 2003. A comparative analysis of evenness index sensitivity. Int. Rev. Hydrobiol. 88(1):3−15.

Belandria, G., J. Asta, and F. Nurit. 1989. Effects of sulphur dioxide and fluoride on ascospore germination of several lichens. Lichenologist 21(1):79−86.

Bergerud, A.T. 1972. Food habits of Newfoundland caribou. J. Wildl. Manage. 36(3):913−923.

Bergman, C.M., and C.J. Krebs. 1993. Diet overlap of collared lemmings and tundra voles at Pearce Point, Northwest Territories. Can. J. Zool. 71:1703−1709.

Beyer, W.N., E.E. Connor, and S. Gerould. 1994. Estimates of soil ingestion by wildlife. J. Wildl. Manage. 58(2):375−382.

Beyer, W.N., J. Spann, and D. Day. 1999. Metal and sediment ingestion by dabbling ducks. Sci. Total Environ. 231:235−239.

Boening, D.W. 1998. Aquatic toxicity and environmental fate of xanthates. Mining Eng. 50(9):65−68.

Boertje, R.D. 1990. Diet quality and intake requirements of adult female caribou of the Denali herd, Alaska. J. Appl. Ecol. 27:420−434.

Borzelleca, J.F., L.W. Condie, and J.L. Egle. 1988. Short-term toxicity (one and 10 day gavage) of barium chloride in male and female rats. J. Am. Coll. Toxicol. 7(5):675−685.

Bowen, H.J.M. 1979. Environmental chemistry of the elements. Academic Press, Inc., New York, NY. (not seen, as cited in ATSDR 1992a)

Bowers, T.S., B.D. Beck, and H.S. Karam. 1994. Assessing the relationship between environmental lead concentrations and adult blood lead levels. Risk Anal. 14:183−189.

Brabets, T.P. 2003. Personal communication (e-mail correspondence to R. Winfree, D. Spirtes, and J. Booth, all with Teck Cominco, Anchorage, AK, dated February 7, 2003, regarding transmittal of water quality data collected at Cape Krusenstern). U.S. Geological Survey, Anchorage, AK.

Brabets, T.P. 2004. Occurrence and distribution of trace elements in snow, streams, and streambed sediment, Cape Krusenstern National Monument, Alaska 2002-2003. USGS Scientific Investigation Report 2004-5229. U.S. Geological Survey.

Brown, Jr., G.E., A.L. Foster, and J.D. Ostergren. 1999. Mineral surfaces and bioavailability of heavy metals: A molecular-scale perspective. Proc. Natl. Acad. Sci. USA 96:3388−3395.

Buck, W.R., R.C. Harris, A.J. Shaw, M.D. Piercey-Normore, A. Tabaee, J. Antonovics, and E.E. Crone. 1999. Unusual lichens under electricity pylons on zinc-enriched soil. Bryologist 102(1):130–132.

Buckner, C.H. 1964. Metabolism, food capacity, and feeding behavior in four species of shrews. Can. J. Zool. 42:259–279.

Cain, B.W., and E.A. Pafford. 1981. Effects of dietary nickel on survival and growth of mallard ducklings. Arch. Environ. Contam. Toxicol. 10:737–745.

Calder, W.A. and E.J. Braun. 1983. Scaling of osmotic regulation in mammals and birds. Am. J. Physiol. 244:R601–R606.

Campbell, K.L., G.L. Weseen, and R.A. MacArthur. 1998. Seasonal changes in water flux, forage intake, and assimilated energy of free-ranging muskrats. J. Wildl. Manage. 62(1):292–299.

Carriere, D., K. Fischer, D. Peakall, and P. Angehrn. 1986. Effects of dietary aluminum in combination with calcium and phosphorous on the ring dove (*Streptopelia risoria*). Water Air Soil Pollut. 30:757–764.

CDC. 2002a. Screening young children for lead poisoning: Guidance for state and local public health officials. CDC Childhood Lead Prevention Program, National Center for Environmental Health, Atlanta, GA.

CDC. 2002b. Managing elevated blood lead levels among young children: Recommendations from the Advisory Committee on Childhood Lead Poisoning Prevention. U.S. Department of Health and Human Services, Centers for Disease Control and Prevention.

Chesemore, D.L. 1975. Ecology of the arctic fox (*Alopex lagopus*) in North America—A review. pp. 143–163. In: The Wild Canids, Their Systematics, Behavioral Ecology and Evolution. M.W. Fox (ed). Van Nostrand Reinhold Company.

Chu, S., S. Tanaka, J.D. Kaunitz, and M.H. Montrose. 1999. Dynamic regulation of gastric surface pH by luminal pH. J. Clin. Invest. 103(5):605–612.

Clark, J. 2005. Personal communication (e-mail to J. Kulas, Teck Cominco Alaska Incorporated, Anchorage, AK, dated March 8, 2005, regarding additional risk assessment notes). Teck Cominco Alaska Incorporated, Anchorage, AK.

Clymo, R.S. 1973. The growth of *Sphagnum*: Some effects of environment. J. Ecol. 61:849–869.

Cominco, RWJ, and PN&D. 1999. DeLong Mountain terminal project, soil/sediment analytical testing data report. Prepared for H.A. Simons, Ltd., Vancouver, BC, Canada. Cominco Alaska Incorporated, RWJ Consulting, and Peratrovich, Nottingham & Drage, Inc., Anchorage, AK.

Connor, J.J., and H.T. Shacklette. 1975. Background geochemistry of some rocks, soils, plants, and vegetables in the conterminous United States. U.S. Geological Survey, Washington, DC. (not seen, as cited in ATSDR 1992a)

Corps. 2001. Pre-dredge sediment characterization, DeLong Mountain Terminal Project, Red Dog, Alaska, August 2000. December 2001. U.S. Army Corps of Engineers, Alaska District.

Corps. 2005. Draft environmental impact statement. Navigation Improvement Delong Mountain Terminal, Alaska. Available at www.poa.usace.army.mil/en/cw/delong/deis.html. Last accessed: August 17, 2006. U.S. Army Corps of Engineers.

Craighead, D.J., and J.J. Craighead. 1987. Tracking caribou using satellite telemetry. Nat. Geog. Res. 3:462–479.

Custer, T.W., and F. Pitelka. 1978. Seasonal trends in summer diet of the Lapland longspur near Barrow, Alaska. Condor 80:295–301.

Custer, T.W., R.G. Osborn, F.A. Pitelka, and J.A. Gessaman. 1986. Energy budget and prey requirements of breeding Lapland longspur near Barrow, Alaska, U.S.A. Arctic Alpine Res. 18(4):415–427.

Dames & Moore. 1983a. Cominco Alaska Inc., environmental baseline studies, Red Dog project. Dames & Moore, Anchorage, AK.

Dames & Moore. 1983b. Supplement to environmental baseline studies, Red Dog project. Prepared for Cominco Alaska, Inc. Dames & Moore, Anchorage, AK.

Daubenmire, R. 1959. A canopy-coverage method of vegetational analysis. Northwest Sci. 33(1):43–64.

Davis, R.D., P.H.T. Beckett, and E. Wollan. 1978. Critical levels of twenty potentially toxic elements in young spring barley. Plant Soil 49:395–408.

Davis, A., R. Barale, G. Brun, R. Forster, T. Gunther, H. Hautefeuille, C.A. van der Heijden, A.G. Knaap, R. Krowke, and T. Kuroki. 1987. Evaluation of the genetic and embryotoxic effects of bix(tri-*n*-butyltin)oxide (TBTO), a broad-spectrum pesticide, in multiple *in vivo* and *in vitro* short-term tests. Muta. Res. 188:65–95.

DEC. 1999. Ecoregions/Assessment Endpoint Project, technical background document for selection and application of default assessment endpoints and indicator species in Alaskan ecoregions. X-0885-04. Available at: www.dec.state.ak.us/spar/csp/guidance/ecoregion_endpoint_report99.pdf. Alaska Department of Conservation, Division of Spill Prevention and Response, Contaminated Sites Program.

DEC. 2000. Risk assessment procedures manual. Alaska Department of Environmental Conservation, Contaminated Sites Remediation Program, June 8, 2000. Alaska Department of Environmental Conservation.

DEC. 2002. Cleanup levels guidance. Alaska Department of Environmental Conservation, Contaminated Sites Remediation Program, November 7, 2002. Alaska Department of Environmental Conservation

DEC. 2003a. Oil and other hazardous substances pollution control. Alaska Department of Environmental Conservation, 18 AAC 75, as amended through January 30, 2003. Alaska Department of Environmental Conservation

DEC. 2003b. Letter from W. Janes to J. Kulas, Teck Cominco Alaska Inc., dated November 26, 2003, regarding final comments – draft DMTS fugitive dust risk assessment work plan. Alaska Department of Environmental Conservation, Juneau, AK.

DEC. 2004a. Letter from R. Sundet to J. Kulas, Teck Cominco Alaska Inc., dated April 16, 2004, regarding comments on the revised DMTS fugitive dust risk assessment work plan dated February 2004. Alaska Department of Environmental Conservation, Anchorage, AK.

DEC. 2004b. Letter from R. Sundet to J. Kulas, Teck Cominco Alaska Inc., dated October 20, 2004, regarding approval of the final DMTS fugitive dust risk assessment work plan dated February 2004 and response to DEC comments on the work plan. Alaska Department of Environmental Conservation, Anchorage, AK.

DEC, Teck Cominco, and Exponent. 2002. Draft fugitive dust background document, DeLong Mountain Regional Transportation System, Alaska. Alaska Department of Environmental Conservation, Anchorage, AK; Teck Cominco Alaska Incorporated, Anchorage, AK; and Exponent, Bellevue, WA.

Denbow, D. M. 2000. Gastrointestinal Anatomy and Physiology. In: Sturkie's Avian Physiology. G. C. Whittow (ed). Academic Press, San Diego, CA.

DFG. 1998a. Catalog of waters important for spawning, rearing or migration of anadromous fishes. Revised November 4, 1998. Alaska Department of Fish and Game, Habitat Division, Juneau, AK.

DFG. 1998b. Methods for aquatic life monitoring to satisfy requirements under 1998 NPDES: NPDES AK-003865-2, Red Dog Mine Site. Alaska Department of Fish and Game, Division of Habitat and Restoration, Fairbanks, AK.

DFG. 2001. Community profile database. Microsoft Access 2000 database. Alaska Department of Fish and Game, Division of Subsistence.

DFG. 2002. River otter. Wildlife Notebook Series. www.state.ak.us/local/akpages/FISH.GAME/notebook/furbear/rvrotter.htm. Accessed on August 16, 2002. Last updated on March 29, 2002. Alaska Department of Fish and Game.

DFG. 2003a. Muskrat. Wildlife Notebook Series. www.state.ak.us/local/akpages/FISH.GAME/notebook/furbear/muskrat.htm. Accessed January 22, 2003. Last updated March 29, 2002. Alaska Department of Fish and Game.

DFG. 2003b. Geese. Wildlife Notebook Series. www.state.ak.us/local/akpages/FISH.GAME/notebook/bird/geese.htm. Accessed January 13, 2003. Last updated March 29, 2002. Alaska Department of Fish and Game.

DFG. 2003c. Caribou management of survey-inventory activities, 1 July 2000-30 June 2002. C. Healy (ed). Alaska Department of Fish and Game, Juneau, AK.

DFG. 2003d. Plovers. Wildlife Notebook Series. www.state.ak.us/local/akpages/FISH.GAME/notebook/bird/plovers.htm. Accessed January 13, 2003. Last updated April 5, 2002. Alaska Department of Fish and Game.

DFG. 2003e. Moose. Wildlife notebook series. www.state.ak.us/adfg/notebook/biggame/moose.htm. Accessed November 3, 2003. Alaska Department of Fish and Game.

Dixon, W.J. 1953. Processing data for outliers. Biometrics. pp. 74–79.

Domingo, J.L., J.L. Paternain, J.M. Llobet, and J. Corbella. 1986. Effects of vanadium on reproduction, gestation, parturition and lactation in rats upon oral administration. Life Sci. 39:819–824.

Dragun, J., and A. Chiasson. 1991. Elements in North American soils. Hazardous Materials Control Resources Institute, Greenbelt, MD.

Drewien, R.C. 1967. Ecological relationships of breeding waterfowl to prairie potholes. P.R. Rep. W-756R-8. South Dakota Department of Game, Fish, and Parks, Pierre, SD. (Not seen, as cited in Granholm 2003.)

Dunning Jr., J.B. 1993. CRC handbook of avian body masses. CRC Press.

E&E. 2002. Wild foods investigation, Northwest Alaska. Public Review Draft Report. Prepared for Alaska Department of Environmental Conservation, Division of Air & Water Quality. Ecology and Environment, Inc.

Eberhardt, L.E. and W.C. Hanson. 1978. Long-distance movements of Arctic foxes tagged in northern Alaska. Can. Field-Nat. 92(4):386–389.

Edens, F.W., E. Benton, S.J. Bursian, and G.W. Morgan. 1976. Effect of dietary lead on reproductive performance in Japanese quail, *Coturnix coturnix japonica*. Toxicol. Appl. Pharmacol. 38:307–314.

Efroymson, R.A., M.E. Will, G.W. Suter, II, and A.C. Wooten. 1997a. Toxicological benchmarks for screening contaminants of potential concern for effects on terrestrial plants: 1997 revision. ES/ER/TM-85/R3. Oak Ridge National Laboratory.

Efroymson, R.A., M.E. Will, and G.W. Suter, II. 1997b. Toxicological benchmarks for screening contaminants of potential concern for effects on soil and litter invertebrates and heterotrophic processes: 1997 revision. ES/ER/TM-126/R2. Oak Ridge National Laboratory.

Eisler, R. 2000. Handbook of chemical risk assessment. Health hazards to humans, plants, and animals. 3 Volumes. Lewis Publishers, Washington, DC.

ENSR. 1990. Port Site sampling program—June 1990. Prepared for Cominco Alaska, Incorporated, Anchorage, AK. ENSR Consulting and Engineering.

ENSR. 1991. Port Site monitoring report—Fall 1990. Prepared for Cominco Alaska, Inc., Anchorage, AK. ENSR Consulting and Engineering, Anchorage, AK.

ENSR. 1992. Red Dog port site monitoring program—September 1991. Document No. 1795-010. Prepared for Cominco Alaska, Inc., Anchorage, AK. ENSR Consulting and Engineering.

ENSR. 1993. Red Dog port site monitoring program. Document No. 1795-015. Prepared for Cominco Alaska, Inc., Anchorage, AK. ENSR Consulting and Engineering.

ENSR. 1996. Red Dog Port Site monitoring program—August 1995. Prepared for Cominco Alaska, Inc., Kotzebue, AK. ENSR Consulting and Engineering.

ept. 1996. Toxicity extrapolation in terrestrial systems. Submitted to California Environmental Protection Agency, Office of Environmental Health, Hazard Assessment, Reproductive and Cancer Assessment Section. ecological planning and toxicology, inc., Corvallis, OR

EVS and Ott Water. 1983. Toxicological, biophysical, and chemical assessment of Red Dog Creek, Delong Mountains, Alaska, 1982. Contract No. AS 46.03.020. Prepared for Alaska Department of Environmental Conservation, Juneau, AK. EVS Environmental Consultants, Inc., North Vancouver, BC, and Ott Water Engineers, Inc., Anchorage, AK.

Exponent. 1999. Ward Cove sediment remediation project, detailed technical studies report. Volume I. Remedial investigation and feasibility study. Prepared for Ketchikan Pulp Company, Ketchikan, AK. Exponent, Bellevue, WA.

Exponent. 2002a. Fugitive dust data report, DeLong Mountain Regional Transportation System, Alaska. Draft. Prepared for Teck Cominco Alaska Incorporated, Anchorage, AK. Exponent, Bellevue, WA.

Exponent. 2002b. Supplemental road sampling and surface material removal verification report. Prepared for Teck Cominco Alaska Incorporated, Anchorage, AK. Exponent, Bellevue, WA.

Exponent. 2002c. Spill site characterization plan, DeLong Mountain Regional Transportation System, Alaska. Draft. Prepared for Teck Cominco Alaska Incorporated, Anchorage, AK. Exponent, Bellevue, WA.

Exponent. 2002d. Concentrate recovery and recycling plan, DeLong Mountain Regional Transportation System, Alaska. Prepared for Teck Cominco Alaska Incorporated, Anchorage, AK. Exponent, Bellevue, WA.

Exponent. 2002e. Evaluation of metals concentrations in caribou tissues. Draft technical memorandum. Prepared for Teck Cominco Alaska Incorporated, Anchorage, AK. Exponent, Bellevue, WA.

Exponent. 2003a. DMTS fugitive dust risk assessment conceptual site model. Prepared for Teck Cominco Alaska Incorporated, Anchorage, AK. Exponent, Bellevue, WA.

Exponent. 2003b. DMTS fugitive dust risk assessment work plan. Draft report. Prepared for Teck Cominco Alaska Incorporated, Anchorage, AK. Exponent, Bellevue, WA.

Exponent. 2003c. Port site characterization data report. Prepared for Teck Cominco Alaska Inc., Anchorage, AK. Exponent, Bellevue, WA.

Exponent. 2003d. Addendum to the concentrate recovery and recycling plan, March 18, 2003. Prepared for Teck Cominco Alaska Incorporated, Anchorage, AK. Exponent, Bellevue, WA.

Exponent. 2003e. Phase I field sampling and analysis plan for the DMTS fugitive dust risk assessment. Prepared for Teck Cominco Alaska Incorporated, Anchorage, AK. Exponent, Bellevue, WA.

Exponent. 2004a. Phase II field sampling and analysis plan for the DMTS fugitive dust risk assessment. Prepared for Teck Cominco Alaska Inc., Anchorage, AK. Exponent, Bellevue, WA.

Exponent. 2004b. Draft DMTS fugitive dust risk assessment work plan. Prepared for Teck Cominco Alaska Inc., Anchorage, AK. Exponent, Bellevue, WA.

Exponent. 2004c. Draft field sampling and analysis plan for supplemental salmonberry and vegetation assessment. Prepared for Teck Cominco Alaska Inc., Anchorage, AK. Exponent, Bellevue, WA.

Exponent. 2004d. Assessment of metals concentrations in salmonberry and sourdock collected near the DMTS. Draft technical memorandum. Prepared for Teck Cominco Alaska Incorporated, Anchorage, AK. Exponent, Bellevue, WA.

Exponent. 2005a. Draft DMTS fugitive dust risk assessment. Prepared for Teck Cominco Alaska Incorporated, Anchorage, AK. Exponent, Bellevue, WA.

Exponent. 2005b. Assessment of metals concentrations in ptarmigan collected near the DMTS. Draft technical memorandum. Prepared for Teck Cominco Alaska Incorporated, Anchorage, AK. Exponent, Bellevue, WA.

Exponent. 2007. Response to U.S. Environmental Protection Agency comments (dated 7 July 2005) on the April 2005 draft DMTS fugitive dust risk assessment. Prepared for Teck Cominco Alaska Incorporated, Anchorage, AK. Exponent, Bellevue, WA.

FDA. No date. Counting calories. Available online at: http://www.fda.gov/fdac/graphics/foodlabelspecial/pg44.pdf. U.S. Food and Drug Administration

FDA. 2003. Human fluid and caloric requirements. Interactive calculator available online at: http://www.fda.gov/cder/cancer/caloricframe.htm. U. S. Food and Drug Administration.

Folkeson, L. 1984. Deterioration of the moss and lichen vegetation in a forest polluted by heavy metals. Ambio 13(1):37–39.

Folkeson, L., and E. Andersson-Bringmark. 1988. Impoverishment of vegetation in a coniferous forest polluted by copper and zinc. Can. J. Bot. 66:417–428.

Ford, S., and L. Hasselbach. 2001. Heavy metals in mosses and soils on six transects along the Red Dog Mine haul road, Alaska. NPS/AR/NRTR-2001/38. National Park Service, Western Arctic National Parklands.

Formigli, L., R. Scelsi, P. Poggi, C. Gregotti, A. DiNucci, E. Sabbioni, L. Gottardi, and L. Manzo. 1986. Thallium-induced testicular toxicity in the rat. Environ. Res. 40:531–539.

Franzmann, A.W., and C.C. Schwartz (eds). 1997. Ecology and management of the North American moose. Smithsonian Institution Press, Washington, DC.

Franzmann, A.W., R.E. Leresche, R.A. Rausch, and J.L. Oldemeyer. 1978. Alaskan moose measurements and weights and measurement-weight relationships. Can. J. Zool. 56:298–306.

Fuller, W.A. 1951. Measurements and weights of northern muskrats. J. Mammal. 32(3):360–362.

Garry, M.R., L.J. Yost, S.S. Shock, and W.J. Shields. 2004. Assessment of metals exposure associated with subsistence use of caribou collected near a mining transport road in northwest Alaska. Poster presented a the Society of Toxicology's 43rd Annual Meeting Baltimore, MD.

Gessaman, J.A. 1972. Bioenergetics of the snowy owl (*Nyctea scandiaca*). Arctic Alpine Res. 4(3):223–238.

Goodman, R.A. (ed). 1997. Update: blood lead levels—United States, 1991–1994. Morbid. Mortal. Weekly Rep. 46(7):141–146.

Gough, L.P. 2003. Personal communication (e-mail correspondence dated March 12, 2003, to T. Hudson, Applied Geology, Inc., regarding Red Dog willow data). U.S. Geological Survey, Anchorage, AK.

Gough, L.P., R.C. Severson, and H.T. Shacklette. 1988. Element concentrations in soils and other surficial materials of Alaska. U.S. Geological Survey Professional Paper 1458. U.S. Geological Survey, Denver, CO.

Gough, L., G.R. Shaver, J. Carroll, D.L. Royer, and J.A. Laundre. 2000. Vascular plant species richness in Alaskan arctic tundra: The importance of soil pH. J. Ecol. 88:54–66.

Goyer, R. 1996. Toxic effects of metals. In: Casarett and Doull's Toxicology: The Basic Science of Poisons. Fifth Edition. C. Klassen, M.O. Amdur, and J. Doull (eds). McGraw-Hill, New York, NY.

Grace, B.W. 1981. A study of threshold sulphur dioxide concentrations for the lichen, *Cladina rangiferina* L. Harm. National Library of Canada.

Granholm. 2003. Green-winged teal (*Anas crecca*): Distribution, abundance, and seasonality. California Wildlife Habitat Relationship System. www.dfg.ca.gov/whdab/B077.html. Accessed February 25, 2003. California Department of Fish and Game, California Interagency Wildlife task group.

Green, R.E., G.J.M. Hirons, and B.H. Cresswell. 1990. Foraging habitats of female common snipe *Gallinago gallinago* during the incubation period. J. Appl. Ecol. 27:325–335.

Gross, W.G., and V.G. Heller. 1946. Chromates in animal nutrition. J. Ind. Hyg. Toxicol. 28(2):52–56.

Hagy, L. 2003. Personal communication (e-mail to S. Shock, Exponent, Bellevue, WA, dated June 25, 2003, regarding spills listed in DEC_Spill_List.xls.) Teck Cominco.

Hall, W. 2003. Personal communication (e-mail to S. Shock, Exponent, Bellevue, WA, dated October 3, 2003, regarding stream sampling data). Teck Cominco.

Hannon, S.J. and G. Dobush. 1997. Pairing status of male willow ptarmigan: Is polygyny costly to males? Anim. Behav. 53:369–380.

Hannon, S.J., P.K. Eason, and K. Martin. 1998. Willow ptarmigan (*Lagopus lagopus*). In: The Birds of North America, No. 369. A. Poole and F. Gill (eds). The Birds of North America, Inc., Philadelphia, PA.

Hanson, W.C., F.W. Whicker, and J.F. Lipscomb. 1975. Lichen forest ingestion rates of free-roaming caribou estimated with fallout cesium-137. pp. 71–79. In: Proc. First Int. Reindeer and Caribou Symposium, 9–11 August 1972, University of Alaska, Fairbanks, Alaska. J.R. Luick, P.C. Lent, D.R. Klein, and R.G. White (eds). Biological Papers of the University of Alaska.

Haseltine, S.D., L. Sileo, D.J. Hoffman, and B.D. Mulhern. 1985. Effects of chromium on reproduction and growth in black ducks. Unpublished data. (Not seen, as cited in Sample et al. 1996.)

Hasselbach, L. 2003. Personal communication (e-mail correspondence transmitting moss data to J. Booth, Teck Cominco, dated March 13, 2003, regarding NPS database). National Park Service, Western Arctic National Parklands.

Hasselbach, L., J.M. Ver Hoef, J. Ford, P. Neitlich, E. Crecelius, S. Berryman, B. Wolk, and T. Bohle. 2005. Spatial patterns of cadmium and lead deposition on and adjacent to National Park Service lands in the vicinity of Red Dog Mine, Alaska. Sci. Total Environ. 348:211–230.

Hawes, M.L. 1977. Home range, territoriality, and ecological separation in sympathetic shrews, *Sorex vagrans* and *Sorex obscurus*. J. Mammal. 58(3):354–367.

Hebert, P.D.N. (ed). 2002. Canada's aquatic environments. www.aquatic.uoguelph.ca. Accessed on January 23, 2003. Last updated 2002. University of Guelph.

Heinz, G. 1974. Effects of low dietary levels of methylmercury on mallard reproduction. Bull. Environ. Contam. Toxicol. 11(4):386–392.

Heinz, G. 1976a. Methylmercury: Second-year feeding effects on mallard reproduction and duckling behavior. J. Wildl. Manage. 40:82–90.

Heinz, G. 1976b. Methylmercury: Second-generation reproductive and behavioral effects on mallard ducks. J. Wildl. Manage 40(4):710–715.

Heinz, G. 1979. Methylmercury: Reproductive and behavioral effects on three generations of mallard ducks. J. Wildl. Manage 43(2):394–401.

Heinz, G.H., D.J. Hoffman, A.J. Krynitsky, and D.M.G. Weller. 1987. Reproduction in mallards fed selenium. Environ. Toxicol. Chem. 6:423–433.

Heinz, G.H., D.J. Hofman, and L.G. Gold. 1989. Impaired reproduction of mallards fed an organic form of selenium. J. Wildl. Manage. 53(2):418–426.

Hemming, J.E. 1987. Annual caribou monitoring report. Prepared for NANA Regional Corporation. Dames & Moore, Anchorage, AK.

Hemming, J.E. 1988. Caribou monitoring report. Prepared for NANA Regional Corporation. Dames & Moore, Anchorage, AK.

Hemming, J.E. 1989. Annual caribou monitoring report. Prepared for NANA Regional Corporation. Dames & Moore, Anchorage, AK.

Hemming, J.E. 1990. Annual caribou monitoring report. Prepared for NANA Regional Corporation. Dames & Moore, Anchorage, AK.

Hemming, J.E. 1991. Annual caribou monitoring report. Prepared for NANA Regional Corporation. Dames & Moore, Anchorage, AK.

Hoefler. 2000. Supplemental submission for the February 1998 application for an amendment to AQC permit to operate No. 9632-AA001 for Red Dog Port, Red Dog, Alaska. Prepared for Teck Cominco Alaska Incorporated, Anchorage, AK.

Holleman, D.F., J.R. Luick, and R.G. White. 1979. Lichen intake estimates for reindeer and caribou during winter. J. Wildl. Manage. 43(1):192–201.

Hudson, R.H., R.K. Tucker, and M.A. Haegele. 1984. Handbook of toxicity to wildlife. Second Edition. Resource Publication No. 153. U.S. Fish and Wildlife Service, Washington, DC.

Hughes, J.H., and E.L. Young, Jr. 1982. Autumn foods of dabbling ducks in southeastern Alaska. J. Wildl. Manage. 46(1):259–263.

Hui, C.A., and W.N. Beyer. 1998. Sediment ingestion of two sympatric shorebird species. Sci. Total Environ. 224:227–233.

Hussell, D.J.T., and R. Montgomerie. 2002. Lapland longspur (*Calcarius lapponicus*). In: The Birds of North America, No. 656. A. Poole and F. Gill (eds). The Birds of North America, Inc., Philadelphia, PA.

ICRP. 1974. Report of the task group on reference man. International Commission on Radiological Protection, New York, NY. (not seen, as cited in ATSDR 1992a)

Ingersoll, C.G., P.S. Haverland, E.L. Brunson, T.J. Canfield, F.J. Dwyer, C.E. Henke, N.E. Kemble, and D.R. Mount. 1996. Calculation and evaluation of sediment effect concentrations for the amphipod *Hyalella azteca* and the midge *Chironomus riparius*. J. Great Lakes Res. 22(3):602–623.

Irving, L. 1960. Birds of Anaktuvuk Pass, Kobuk, and Old Crow. A study in arctic adaptation. U.S. National Museum Bulletin 217. Smithsonian Institution, Washington, DC. 409 pp.

Jackson, N., S.W. Gibson, and M.H. Stevenson. 1986. Effects of short-and long-term feeding of zinc-oxide-supplemented diets on the mature, female domestic fowl with special reference to tissue mineral content. Brit. J. Nutr. 55:333–349.

Jarvinen, A.W., and G.T. Ankley. 1999. Linkage of effects to tissue residues: Development of a comprehensive database for aquatic organisms exposed to inorganic and organic chemicals. SETAC Technical Publication Series. Society of Environmental Toxicology and Chemistry, Pensacola, FL.

Johnson, D., A.L. Mehring, and H.W. Titus. 1960. Tolerance of chickens for barium. Proc. Soc. Exp. Biol. Med. 104:436–438.

Johnson, K. 1995. Green-winged teal (*Anas crecca*). In: The Birds of North America, No. 193. A. Poole and F. Gill (eds). The Academy of Natural Sciences, Philadelphia, PA, and The American Ornithologists' Union, Washington, DC.

Kabata-Pendias, A., and H. Pendias. 1992. Trace Elements in soils and plants. Second Edition. CRC Press, Inc., Boca Raton, FL.

Kålås, J.A., T.H. Ringsby, and S. Lierhagen. 1995. Metals and selenium in wild animals from Norwegian areas close to Russian nickel smelters. Environ. Monitor. Assess. 36:251–270.

Kashulina, G., C. Reimann, and D. Banks. 2003. Sulphur in the Arctic environment (23): Environmental impact. Environ. Pollut. 124:151−171.

Kelley, K.D., and T. Hudson. 2003. The natural dispersal of metals to the environment in the Wulik-Ikalukrok River area, western Brooks Range, Alaska. U.S. Geological Survey, Denver, CO, and Applied Geology, Inc., Sequim, WA.

Kerlinger, P., and M.R. Lein. 1988. Causes of mortality, fat condition, and weights of wintering snowy owls. J. Field Ornithol. 59(1):7−12.

Kulas, J. 2003. Personal communication (e-mail to S. Shock, Exponent, Bellevue, WA, dated June 17, 2003, regarding revised draft field sampling and analysis plan). Teck Cominco.

Kulas, J. 2004. Personal communication (teleconference with S. Shock, Exponent, Bellevue, WA, on Jan 5, 2004, regarding former spills). Teck Cominco.

Kulas, J. 2005. Personal communication (e-mail to S. Shock, Exponent, Bellevue, WA, dated March 8, 2005, regarding comments on risk assessment). Teck Cominco, Alaska Incorporated, Anchorage, AK.

Lambin, X., C.J. Krebs, and B. Scott. 1992. Spacing system of the tundra vole (*Microtus oeconomus*) during the breeding season in Canada's western Arctic. Can. J. Zool. 70:2068−2072.

Lamprecht, R., and W. Graber. 1996. Modeling dry deposition of dust along the Dalton Highway. Ecol. Studies 120:325−345.

Langmuir, D., P Chrostowski, B. Vigneault, and R. Chaney. 2004. Issue paper on the environmental chemistry of metals. Submitted to U.S. Environmental Protection Agency, Risk Assessment Forum, Washington, DC. ERG, Lexington, MA.

Laskey, J.W., and F.W. Edens. 1985. Effects of chronic high-level manganese exposure on male behavior in the Japanese quail (*Coturnix coturnix japonica*). Poult. Sci. 64:579−584.

Laskey, J.W., G.L. Rehnberg, J.F. Hein, and S.D. Carter. 1982. Effects of chronic manganese (Mn_3O_4) exposure on selected reproductive parameters in rats. J. Toxicol. Environ. Health 9:677−687.

Laws, E.P. 1994. Revised interim soil lead guidance for CERCLA sites and RCRA corrective action facilities. Letter to Regional Administrators I-X. U.S. EPA OSWER Directive #9355.4-12. U.S. Environmental Protection Agency, Washington, DC.

Lent, P.C. 1966. The caribou of northwestern Alaska. pp. 481−517. In: Environment of the Cape Thompson Region, Alaska. N.J. Wilimovsky and J.N. Wolfe (eds). U.S. Atomic Energy Commission, Division of Technical Information, Oak Ridge, TN.

Lepore, P.D., and R.F. Miller. 1965. Embryonic viability as influenced by excess molybdenum in chicken breeder diets. Proc. Soc. Exp. Biol. Med. 118:155−157.

Liblik, V., and M. Pensa. 2001. Specifics and temporal changes in air pollution in areas affected by emissions from oil shale industry, Estonia. Water Air Soil Pollut. 130:1787–1792.

Long, E.R., D.D. MacDonald, S.L. Smith, and F.D. Calder. 1995. Incidence of adverse biological effects within ranges of chemical concentrations in marine and estuarine sediments. Environ. Manage. 19(1):81–97.

MacDonald, S.O., and J.A. Cook. 2002. Mammal inventory of Alaska's National Parks and Preserves, Northwest Network: Western Arctic Parklands. Bering Land Bridge National Preserve, Cape Krusenstern National Monument, Kobuk Valley National Park Noatak National Preserve. Annual Report 2001. University of Alaska Museum.

MacDonald, D.D., C.G. Ingersoll, and T.A. Berger. 2000. Development and evaluation of consensus-based sediment quality guidelines for freshwater ecosystems. Arch. Environ. Contam. Toxicol. 39:20–31.

Mackenzie, R.D., R.U. Byerrum, C.F. Decker, C.A. Hoppert, and R.F. Langham. 1958. Chronic toxicity studies. II: hexavalent and trivalent chromium administered in drinking water to rats. Am. Med. Assoc. Arch. Ind. Health 18:232–234.

Maddaloni, M., P. Goodrum, G. Diamond, W. Manton, C. Blum, N. Lolacono, and J. Graziano. 1998. Meal-weighted bioavailability of soil-borne lead in adults: Use of empirical data analyzed by stable isotope dilution, and probabilistic worker exposure scenarios. Poster Presentation at the 1998 Society of Toxicology Annual Meeting, Seattle, WA.

Martell, A.M., and A.M. Pearson. 1978. The small mammals of the Mackenzie Delta Region, Northwest Territories, Canada. Arctic 31(4):475–488.

McBride, M.B. 1994. Environmental chemistry of soils. Oxford University Press, Inc., New York, NY. 406 pp.

McLelland, J. 1979. Digestive System. In: Form and function in birds. Academic Press, London. 459 pp.

McCune, B. 1988. Lichen communities along O_3 and SO_2 gradients in Indianapolis. Bryologist 91:224–228.

McGeer, J., G. Hennington, R. Lanno, N. Fisher, K, Sappington, and J. Drexler. 2004. Issue Paper on the Bioavailability and Bioaccumulation of Metals. Submitted to USEPA, Risk Forum, Washington, D.C. by Eastern Research Group (ERG), Lexington, MA.

Mehring, A.L., J.H. Brumbaugh, A.J. Sutherland, and H.W. Titus. 1960. The tolerance of growing chickens for dietary copper. Poult. Sci. 39:713–719.

Merrill, A.L., and B.K. Watt. 1973. Energy values of food: basis and derivation. Agricultural Handbook No. 74. U.S. Department of Agriculture, Human Nutrition Research Branch.

Miller, D.R. 1976. Biology of the Kaminuriak population of barren-ground caribou. Part 3: Taiga winter range relationships and diet. Canadian Wildlife Services Report Series Number 36.

Moitoret, C. no date. Personal communication. (not seen, as cited by Paulson 1995)

Morris, W.A., and A.G. Ott. 2001. Metals concentrations in juvenile Dolly Varden (*Salvelinus malma*) sampled at two streams along the DeLong Mountain Regional Transportation System, Red Dog Mine. Alaska Department of Fish and Game.

Mueller, H. 1999. Common snipe (*Gallinago gallinago*). In: The Birds of North America, No. 417. A. Poole and F. Gill (eds). The Birds of North America, Inc., Philadelphia, PA.

Mueller, K.A., E. Snyder-Conn, and T. Doyle. 1993. Contaminant baseline data for water, sediments, and fish of Selawik National Wildlife Refuge, Alaska, 1987–1988. Technical Report No. NAES-TR-93-02. Fish and Wildlife Service, U.S. Department of Interior, Fairbanks, AK.

Nagy, K.A. 1987. Field metabolic rate and food requirement scaling in mammals and birds. Ecol. Monogr. 57(2):111−128.

Nagy, K.A., I.A. Girard, and T.K. Brown. 1999. Energetics of free-ranging mammals, reptiles, and birds. Annu. Rev. Nutr. 19:247−277.

Nash III, T.H. 1972. Simplification of the Blue Mountain lichen communities near a zinc factory. Bryologist 75:315−324.

Nash, III, T.H., and C. Gries. 2002. Lichens as bioindicators of sulfur dioxide. Symbiosis 33:1−21.Nation, J.R., A.E. Bourgeois, D.E. Clark, and M.F. Hare. 1983. The effects of chronic cobalt exposure on behavior and metallothionein levels in the adult rat. Neurobehav. Toxicol. Teratol. 5:9–15.

Nation, J.R., A.E. Bourgeois, D.E. Clark, and M.F. Hare. 1983. The effects of chronic cobalt exposure on behavior and metallothionein levels in the adult rat. Neurobehav. Toxicol. Teratol. 5:9−15.

Neal, T.J. 1968. A comparison of two muskrat populations. Iowa State J. Sci. 43:193−210. (not seen, as cited in U.S. EPA 1993)

Nemec, M.D., J.F. Holson, C.H. Farr, and R.D. Hood. 1998. Developmental toxicity assessment of arsenic acid in mice and rabbits. Reprod. Toxicol. 12(6):647−658.

Newman, M.C., G.L. Diamond, C. Menzie, J. Moya, and J. Nriagu. 2003. Issue paper on metal exposure assessment. Draft. Submitted to U.S. Environmental Protection Agency, Risk Assessment Forum, Washington, DC. ERG, Lexington, MA.

Nissen, L.R., and N.W. Lepp. 1997. Baseline concentrations of copper and zinc in shoot tissues of a range of *Salix* species. Biomass Bioenergy 12(2):115−120.

Nobmann, E.D., T. Byers, A.P. Lanier, J.H. Hankin, and M.Y. Jackson. 1992. The diet of Alaska Native adults: 1987–1888. Am. J. Clin. Nutr. 55:1024–1032.

Norris, R.H. and A. George. 1993. Analysis and interpretation of benthic macroinvertebrate surveys. Chapter 7. In: Freshwater Biomonitoring and Benthic Macroinvertebrates. D.M. Rosenberg and V.H. Resh (eds). Chapman & Hall, New York, NY.

NTP. 1994. Technical report on the toxicology and carcinogenesis studies of barium chloride dihydrate (CAS No. 10326-27-9) in F344/N rats and B6C3F1 mice (drinking water studies). NTP TR 432. NIH Publication No. 94-3163. U.S. Department of Health and Human Services, Public Health Service, National Institutes of Health, National Toxicological Program, Research Triangle Park, NC.

Nutrition Data. 2006. Nutrition facts for Caribou Bone Marrow. Available at: http://www.nutritiondata.com/facts-B0000101c226S.html. Last accessed: October 5, 2006.

Ondreicka, R., E. Ginter, and J. Kortus. 1966. Chronic toxicity of aluminium in rats and mice and its effects on phosphorus metabolism. Brit. J. Ind. Med. 23:305–312.

Ott, A.G. 2004. Aquatic biomonitoring at Red Dog Mine, 2003 National Pollution Discharge Elimination System Permit No. AK-003865-2. Technical Report No. 04-02. Alaska Department of Natural Resources, Office of Habitat Management and Permitting, Fairbanks, AK.

Ott, A.G., and W.A. Morris. 2004. Juvenile Dolly Varden whole body metals analyses, Red Dog Mine (2002). Technical Report No. 04-01. Alaska Department of Natural Resources, Office of Habitat Management and Permitting.

Parmelee, D. 1992. Snowy owl (*Nyctea scandiaca*). In: The Birds of North America, No. 10. A. Poole, P. Stettenheim, and F. Gill (eds). The Academy of Natural Sciences, Philadelphia, PA, and The American Ornithologists' Union, Washington, DC.

Pattee, O.H. 1984. Eggshell thickness and reproduction in American kestrels exposed to chronic dietary lead. Arch. Environ. Contam. Toxicol. 13:29–34.

Pattee, O.H., S.N. Wiemeyer, and D.M. Swineford. 1988. Effects of dietary fluoride on reproduction in eastern screech-owls. Arch. Environ. Contam. Toxicol. 17:213–218.

Paulson, D.R. 1995. Black-bellied plover (*Pluvialis squatarola*). In: The Birds of North America, No. 186. A. Poole and F. Gill (eds). The Academy of Natural Sciences, Philadelphia, PA, and The American Ornithologists' Union, Washington, DC.

Peek, J.M. 1974. A review of moose food habits studies in North America. Naturaliste Can. 101:195–215.

Peraza, M.A., F. Ayala-Fierro, D.S. Barber, E. Casarez, and L.T. Rael. 1998. Effects of micronutrients on metal toxicity. Environ. Health Perspect. 106(Suppl 1):203–216.

Perry, H.M., E.F. Perry, M.N. Erlanger, and S.J. Kopp. 1983. Cardiovascular effects of chronic barium ingestion, In: Proc. 17th Ann. Conf. Trace Substances in Environ. Health, Vol. 17. University of Missouri Press, Columbia, MO.

Pielou, E.C. 1966. The measurement of diversity in different types of biological collections. J. Theoret. Biol. 13:131–144.

Pirkle, J.L., D.J. Brody, E.W. Gunter, R.A. Kramer, D.C. Paschal, K.M. Flegal, and T.D. Matte. 1994. The decline in blood lead levels in the United States: The National Health and Nutrition Examination Surveys (NHANES). J. Am. Med. Assoc. 272(4):284–291.

Pitelka, F.A., P.Q. Tomich, and G.W. Treichel. 1955. Ecological relations of jaegers and owls as lemming predators near Barrow, Alaska. Ecol. Monogr. 25(3):85–117.

Pocock, S.J., M. Smith, and P. Baghurst. 1994. Environmental lead and children's intelligence: A systematic review of the epidemiological evidence. Br. Med. J. 309:1189–1197.

Pollard, R.H. 1994a. 1993 annual caribou monitoring report. Prepared for NANA Regional Corporation. LGL Alaska Research Associates, Inc., Anchorage, AK.

Pollard, R.H. 1994b. 1994 annual caribou monitoring report. Prepared for NANA Regional Corporation. LGL Alaska Research Associates, Inc., Anchorage, AK.

Rasanen, L. 1979. Nutrition survey of Finnish rural children. VI. Methodological study comparing the 24-hour recall and the dietary history interview. Am. J. Clin. Nutr. 32(12):2560–2567.

Reed, A., D.H. Ward, D.V. Derksen, and J.S. Sedinger. 1998. Brant (*Branta bernicla*). In: The Birds of North America, No. 337. A. Poole and F. Gill (eds). The Birds of North America, Inc., Philadelphia, PA.

Remington, T.E., and C.E. Braun. 1988. Carcass composition and energy reserves of sage grouse during winter. Condor 90(1):15–19.

Richardson, D.H.S. 1992. Pollution monitoring with lichens. Richmond Publishing Co. Ltd., Slough, England. Naturalists' Handbook 19.

Richardson, D.H.S., and E. Nieboer. 1983. Ecophysiological responses of lichens to sulphur dioxide. J. Hattori Bot. Lab. No. 54:331–351.

Risenhoover, K.L. 1989. Composition and quality of moose winter diets in interior Alaska. J. Wildl. Manage. 53(3):568–577.

Rosenfeld, I., and O.A. Beath. 1954. Effect of selenium on reproduction in rats. Proc. Soc. Exper. Biol. Med. 87(2):295–297.

RWJ. 1997. Red Dog port site monitoring program. Prepared for Cominco Alaska, Inc., Anchorage, AK. RWJ Consulting, Chugiak, AK.

RWJ. 1998. Red Dog Mine vegetation and soil monitoring program. Final Report to Cominco Alaska Incorporated, Kotzebue, AK. RWJ Consulting, Chugiak, AK.

RWJ. 2001. DeLong Mountain terminal 2000 environmental studies. Final Report. Prepared for Teck Cominco Alaska Incorporated. RWJ Consulting.

Salemaa, M., I. Vanha-Majamaa, and J. Derome. 2001. Understorey vegetation along a heavy-metal pollution gradient in SW Finland. Environ. Pollut. 112:339−350.

Sample, B.E., and C.A. Arenal. 1999. Allometric models for interspecies extrapolation of wildlife toxicity data. Bull. Environ. Contam. Toxicol. 62:653−663.

Sample, B.E., D.M. Opresko, and G.W. Suter, II. 1996. Toxicological benchmarks for wildlife: 1996 revision. ES/ER/TM-86/RS. Prepared for the U.S. Department of Energy, Office of Environmental Management. Oak Ridge National Laboratory, Risk Assessment Program, Health Sciences Research Division, Oak Ridge, TN.

Scannell, P.W. and A.G. Ott. 2001. Aquatic biomonitoring at Red Dog Mine, 2000, National Pollution Discharge Elimination System Permit No. AK-003865-2. Technical Report No. 04-02. Alaska Department of Fish and Game, Habitat and Restoration Division, Fairbanks, AK.

Scannell, P.W. 2005. Maniilaq fish tissue data, Wulik and Noatak Rivers. Prepared for Alaska Department of Natural Resources Office of Habitat Management and Permitting. Scannell Technical Services, Schodack Landing, NY.

Scannell, P.W. and A.G. Ott. 2006. Aquatic Baselines Sampling, Wulik River Drainage. Volume I and II: Summary of Biological and Water Quality Information. Technical Report No. 03-05. Alaska Department of Natural Resources Office of Habitat Management and Permitting.

Schlatterer, B., T.M.M. Coenen, E. Ebert, R. Grau, V. Hilbig, and R. Munk. 1993. Effects of bis(tri-*n*-butyltin)oxide in Japanese quail exposed during egg laying period: An interlaboratory comparison study. Arch. Environ. Contam. Toxicol. 24:440–448.

Schlicker, S.A., and D.H. Cox. 1968. Maternal dietary zinc, and development and zinc, iron, and copper content of the rat fetus. J. Nutr. 95:287–294.

Schroeder, M. 1998. Birds of the northwest Alaska areas. www.npwrc.usgs.gov/resource/othrdata/chekbird/r7/noalaska.htm. Accessed on October 16, 2002. Last updated on September 1, 1998. National Park Service, Jamestown, ND.

Schroeder, H.A., and M. Mitchener. 1971. Toxic effects of trace elements on the reproduction of mice and rats. Arch. Environ. Health. 23:102−106.

Schroeder, H.A., M. Mitchener, and A.P. Nason. 1970. Zirconium, niobium, antimony, vanadium and lead in rats: Life term studies. J. Nutr. 100:59−68.

Schwartz, C.C., W.L. Regelin, and A.W. Franzmann. 1984. Seasonal dynamics of food intake in moose. ALCES 20:223–244.

Schwartz, J. 1995. Lead, blood pressure, and cardiovascular disease in men. Arch. Environ. Health 50(1):31−37.

Scotter, G. 1967. The winter diet of barren-ground caribou in northern Canada. Can. Field-Nat. 81:33−39.

Seastedt, T.R., and S.F. MacLean. 1979. Territory size and composition in relation to resource abundance in Lapland longspurs breeding in Arctic Alaska. Auk 96:131−142.

Seaward, M.R.D. 1995. Use and abuse of heavy metal bioassays in environmental monitoring. Sci. Tot. Environ. 176:129−134.

Sharma, J., R.L. Sharma, H.B. Singh et al. 1986. Hazards and analysis of thallium—a review. Toxicol. Environ. Chem. 11:93−116. (Not seen, as cited in ATSDR 1992c.)

Shock, S.S., B.A. Bessinger, Y.W. Lowney, and J.L. Clark. 2007. Assessment of the solubility and bioaccessibility of barium and aluminum in soils affected by mine dust deposition. Environ. Sci. Technol. 41(13):4813–4820.

Silva, M., and J.A. Downing. 1995. Handbook of mammalian body masses. CRC Press, Boca Raton, FL. 359 pp.

Skoryna, S.C. 1981. Effects of oral supplementation with stable strontium. Can. Med. Assoc. J. 125:703−712.

Spatt, P.D., and M.C. Miller. 1981. Growth conditions and vitality of *Sphagnum* in a tundra community along the Alaska pipeline haul road. Arctic 34(1):48−54.

Stahl, J.L., J.L. Greger, and M.E. Cook. 1990. Breeding-hen and progeny performance when hens are fed excessive dietary zinc. Poult. Sci. 69:259−263.

Stanek III, E.J., and E.J. Calabrese. 2000. Daily soil ingestion estimates for children at a Superfund site. Risk Anal. 20:627–635.

Stanley, T.R., J.W. Spann, G.J. Smith, and R. Rosscoe. 1994. Main and interactive effects of arsenic and selenium on mallard reproduction and duckling growth and survival. Arch. Environ. Contam. Toxicol. 26:444–451.

Stevens, C.E., and I.D. Hume. 1995. Comparative physiology of the vertebrate digestive system. Cambridge University Press. 400 pp.

Stimmelmayr, R. 1994. Ecology of reindeer on Hagemeister Island, Alaska. Master Thesis. University of Alaska Fairbanks.

Sundet, R. 2002a. Personal communication (letter to E. Howarth, Village of Noatak, Noatak, AK, dated August 22, 2002, regarding transmittal of information from Red Dog Mine public meeting on June 5, 2002, in Noatak, AK). Alaska Department of Environmental Conservation, Juneau, AK.

Sundet, R. 2002b. Personal communication (letter to J. Norton, IRA, Kivalina, AK, dated August 28, 2002, regarding transmittal of information from Red Dog Mine public meeting on June 17, 2002, in Kivalina, AK). Alaska Department of Environmental Conservation, Juneau, AK.

Sutou, S., K. Yamamoto, H. Sendota, and M. Sugiyama. 1980. Toxicity, fertility, teratogenicity, and dominant lethal tests in rats administered cadmium subchronically. II. Fertility, teratogenicity, and dominant lethal tests. Ecotoxicol. Environ. Saf. 4:51–56.

Takenaka, S., H. Oldiges, H. Konig, D. Hochrainer, and G. Oberdoerster. 1983. Carcinogenicity of cadmium aerosols in Wistar rats. J. Natl. Cancer Inst. 70:367–373.

Teck Cominco. 2002. Map provided by Teck Cominco Alaska Incorporated, Anchorage, AK.

Teck Cominco. 2003a. 2003 port site supplemental characterization data report, DeLong Mountains Regional Transportation System, Alaska. Prepared by Teck Cominco Alaska Incorporated, Anchorage, Alaska. Unpublished report.

Teck Cominco. 2003b. Lead concentrate material safety data sheet. Teck Cominco Metals Ltd., Vancouver, BC.

Teck Cominco. 2003c. Memorandum from K. Turner to Teck Cominco Alaska Incorporated, Red Dog Mine, dated May 20, 2003, regarding concentrate spill site recovery and restoration report, spill site SP03, April 3–4, 2003. Teck Cominco Alaska Incorporated, Anchorage, AK.

Teck Cominco. 2003d. Memorandum from K. Turner to Teck Cominco Alaska Incorporated, Red Dog Mine, dated May 20, 2003, regarding concentrate spill site recovery and restoration report, spill site SP05, April 3–4, 2003. Teck Cominco Alaska Incorporated, Anchorage, AK.

Teck Cominco. 2003e. Report on the 2002 spill site characterization sampling program, sampling procedures and summary of data collected, DeLong Mountains Regional Transportation System, Alaska. Draft. Teck Cominco Alaska Incorporated, Anchorage, AK.

Teck Cominco. 2003f. Zinc concentrate material safety data sheet. Teck Cominco Metals Ltd., Vancouver, BC.

Teck Cominco. 2004a. 2002–2003 DMTS concentrate spill site characterization report, spill site SP-10. Teck Cominco Alaska Incorporated, Anchorage, AK.

Teck Cominco. 2004b. 2002–2003 DMTS concentrate spill site characterization report, spill site SP-13. Teck Cominco Alaska Incorporated, Anchorage, AK.

Teck Cominco. 2004c. 2002–2003 DMTS concentrate spill site characterization report, spill site SP-15. Teck Cominco Alaska Incorporated, Anchorage, AK.

Teck Cominco. 2004d. 2002 DMTS concentrate spill site characterization report, spill site SP-17. Teck Cominco Alaska Incorporated, Anchorage, AK.

Teck Cominco. 2004e. 2002–2003 DMTS concentrate spill site characterization report, spill site SP-27. Teck Cominco Alaska Incorporated, Anchorage, AK.

Teck Cominco. 2004f. 2002–2003 DMTS concentrate spill site characterization report, spill site SP-29. Teck Cominco Alaska Incorporated, Anchorage, AK.

Teck Cominco. 2004g. 2002–2003 DMTS concentrate spill site characterization report, spill site SP-30. Teck Cominco Alaska Incorporated, Anchorage, AK.

Teck Cominco. 2004h. Annual air quality data report, village air monitoring, Kivalina, Alaska, October 1, 2003–September 30, 2004. Project 30-06. Teck Cominco Alaska Incorporated, Anchorage, AK.

Teck Cominco. 2004i. Annual air quality data report, village air monitoring, Noatak, Alaska, April 1, 2003–March 31, 2004. Project 30-06. Teck Cominco Alaska Incorporated, Anchorage, AK.

Teck Cominco. 2004k. DeLong Mountain Regional Transportation System port facility, public access control plan. Teck Cominco Alaska Incorporated. Anchorage, AK.

Teck Cominco. 2004j. Map provided by Teck Cominco Alaska Incorporated. Anchorage, AK.

Teck Cominco. 2005a. Delong Mountain Transportation System road historic concentrate spill site closeout reports for: SP-01, SP-02, SP-04, SP06, SP-08, SP-09, SP-11, SP-16 and 26, SP-18, SP-19, SP-21, SP-22, SP-24, SP-25, and SP-28. Prepared for Alaska Department of Environmental Conservation, Fairbanks, AK. Teck Cominco Alaska Incorporated, Anchorage, AK.

Teck Cominco. 2005b. Summary of mine related fugitive dust studies, Red Dog Mine site. Teck Cominco Alaska Incorporated, Anchorage, AK.

Teck Cominco. 2007a. Environmental management plan 2007: Mine site CSB fugitives reduction. Available at: www.dec.state.ak.us/air/doc/RD_EMP-2007.pdf. Accessed June 27, 2007. Teck Cominco Alaska Incorporated, Anchorage, AK.

Teck Cominco. 2007b. Mineral weathering in Red Dog soils. Report 2007RR01. Available at: www.dec.state.ak.us/air/doc/RD_min_weather_soils-feb07.pdf. Accessed June 27, 2007. Teck Cominco Alaska Incorporated, Anchorage, AK.

Teck Cominco. 2007c. Mineral weathering in Red Dog soils: Leaching. Report 2007RR06. Available at: www.dec.state.ak.us/air/doc/RD_soil_leach-feb07.pdf. Accessed June 27, 2007. Teck Cominco Alaska Incorporated, Anchorage, AK.

Teck Cominco. 2007d. Red Dog Mine dust update—May 11, 2007. Available at: www.dec.state.ak.us/air/doc/RD_Mine_Dust_May07.pdf. Accessed June 27, 2007. Teck Cominco Alaska Incorporated, Anchorage, AK.

Tetra. 1998. Premium anhydrous calcium chloride product data sheet. Available at: www.tetratec.com/ business_units/calcium_chloride/data/exp ress.html. Last updated 1998. Accessed September 2, 2004. Tetra Chemicals, Houston, TX.

Terres, J.K. 1980. The Audubon Society Encyclopedia of North American Birds. Alfred A. Knopf, New York, NY.

Thun, M.J., T.M. Schnorr, A.B. Smith and W.E. Halperin. 1985. Mortality among a cohort of U.S. cadmium production workers: An update. J. Natl. Cancer Inst. 74(2):325–333.

Tuck, L.M. 1972. The snipes: A study of the genus *Capella*. Canadian Wildlife Service Monograph Series—Number 5. Environment Canada, Wildlife Service.

Tyler, G. 1989. Uptake, retention and toxicity of heavy metals in lichens. Water Air Soil Pollut. 47:321–333.

U.S. EPA. 1984. Draft environmental impact statement, Red Dog Mine project northwest Alaska. EPA 910/9-84-122a. Volume I and II. U.S. Environmental Protection Agency.

U.S. EPA. 1986. Subchronic (90-day) toxicity of thallium sulfate in Sprague-Dawley rats. Office of Solid Waste, Washington, DC. (Not seen, as cited in U.S. EPA 2005.)

U.S. EPA. 1988. Recommendations for and documentation of biological values for use in risk assessment. EPA/600/6-87/008. U.S. Environmental Protection Agency, Office of Research and Development, Cincinnati, OH.

U.S. EPA. 1989. Risk assessment guidance for Superfund. Volume I: Human Health Evaluation Manual (Part A). Interim Final Report. EPA/540/1-89/002. U.S. Environmental Protection Agency, Office of Emergency and Remedial Response, Washington, DC.

U.S. EPA. 1991. Risk assessment guidance for Superfund. Volume I: Human health evaluation manual supplemental guidance. Standard default exposure factors. Interim Final. OSWER Directive 9285.6-03. U.S. Environmental Protection Agency, Office of Emergency and Remedial Response, Washington, DC.

U.S. EPA. 1992a. Dermal exposure assessment: Principles and applications. Interim Report. EPA 600/8-91-011B. U.S. Environmental Protection Agency.

U.S. EPA. 1992b. Supplemental guidance to RAGS: Calculating the concentration term. OSWER Directive 9285.7-081. U.S. Environmental Protection Agency.

U.S. EPA. 1993. Wildlife exposure factors handbook. Volumes I and II. EPA/600/R-93/187. U.S. Environmental Protection Agency, Office of Research and Development, Washington, DC.

U.S. EPA. 1994. Guidance manual for the integrated exposure uptake biokinetic model for lead in children. EPA/540/R-93/081. U.S. Environmental Protection Agency, Technical Review Workgroup for Lead.

U.S. EPA. 1995a. Great Lakes water quality initiative technical support document for wildlife criteria. EPA-820-B-95-009. U.S. Environmental Protection Agency, Office of Water, Washington, DC.

U.S. EPA. 1995b. Great Lakes water quality initiative criteria documents for the protection of wildlife. EPA-820-B-95-008. U.S. Environmental Protection Agency, Office of Water, Washington, DC.

U.S. EPA. 1996a. Soil screening guidance: User's guide. U.S. Environmental Protection Agency, Office of Solid Waste and Emergency Response, Washington, DC.

U.S. EPA. 1996b. Soil screening guidance: Technical background document. EPA 540/R-95/128. U.S. Environmental Protection Agency, Office of Solid Waste and Emergency Response, Washington, DC.

U.S. EPA. 1996c. Recommendations of the Technical Review Workgroup for Lead for an interim approach to assessing risks associated with adult exposures to lead in soil. U.S. Environmental Protection Agency, Technical Review Workgroup for Lead.

U.S. EPA. 1997a. Ecological risk assessment guidance for Superfund: Process for designing and conducting ecological risk assessments. Interim Final. U.S. Environmental Protection Agency, Environmental Response Team, Edison, NJ.

U.S. EPA. 1997b. Exposure factors handbook. EPA/600/P-95/002Fa. U.S. Environmental Protection Agency.

U.S. EPA. 1997c. Mercury study report to congress. Volume III: Fate and transport of mercury in the environment. Office of Air Quality Planning and Standards and Office of Research and Development, U.S. Environmental Protection Agency. EPA-452/R-97-005. December 1997.

U.S. EPA. 1998. Guidance for identifying pesticide chemicals that have a common mechanism of toxicity, for use in assessing the cumulative toxic effects of pesticides. Office of Pesticide Programs, Office of Prevention, Pesticides, and Toxic Substances, Washington, DC.

U.S. EPA. 1999a. Risk assessment guidance for Superfund. Volume 3. Part A, Process for conducting probabilistic risk assessment. Draft. Revision 5. U.S. Environmental Protection Agency, Office of Emergency and Remedial Response.

U.S. EPA. 1999b. Short sheet: IEUBK model bioavailability variable. EPA/540/F-00/006. OSWER #9285.7-32. U.S. Environmental Protection Agency, Office of Solid Waste and Emergency Response.

U.S. EPA. 1999c. Toxicological review, cadmium and compounds (CAS No. 7440-43-9). Available at: http://www.epa.gov/ncea/pdfs/cad/cad1.pdf. U.S. Environmental Protection Agency, Office of Research and Development, National Center for Environmental Assessment.

U.S. EPA. 1999d. Rapid bioassessment protocols for use in streams and wadeable rivers: Periphyton, benthic macroinvertebrates, and fish. Technical Report No. 841-B-99-002. U.S. Environmental Protection Agency, Office of Wetlands, Oceans, and Watersheds.

U.S. EPA. 2002a. Blood lead concentrations of U.S. adult females: Summary statistics from Phases 1 and 2 of the National Health and Nutrition Evaluation Survey (NHANES III). U.S. Environmental Protection Agency, Technical Review Workgroup for Lead.

U.S. EPA. 2002b. Calculating exposure point concentrations at hazardous waste sites. Draft. OSWER 9285.6-10. U.S. Environmental Protection Agency, Office of Emergency and Remedial Response, Washington, DC.

U.S. EPA. 2002c. National recommended water quality criteria: 2002. EPA 822-R-02-047. U.S. Environmental Protection Agency, Office of Water, Washington, DC.

U.S. EPA. 2002d. Calculating upper confidence limits for exposure point concentrations at hazardous waste sites. OSWER 9285.6-10. December 2002. U.S. Environmental Protection Agency, Office of Solid Waste and Emergency Response, Washington, DC.

U.S. EPA. 2003a. Ecological soil screening levels for barium. Interim final. OSWER Directive 9285.7063. U.S. Environmental Protection Agency, Office of Solid Waste and Emergency Response, Washington, DC.

U.S. EPA. 2003b. Issue paper on metal exposure assessment. Draft. Submitted to U.S. Environmental Protection Agency by M.C. Newman, G.L. Diamond, C. Menzie, J. Moya, and J. Nriagu.

U.S. EPA. 2003c. Region 9 preliminary remediation goals (PRGs) table. Available at www.epa.gov/region09/waste/sfund/prg/index.htm. U.S. Environmental Protection Agency, Region 9.

U.S. EPA. 2003d. Assessing intermittent or variable exposures at lead sites. OSWER 9285.7-76. EPA-540-R-03-008. U.S. Environmental Protection Agency, Office of Solid Waste and Emergency Response, Washington, DC.

U.S. EPA. 2003e. Evaluation of dermal contact and inhalation exposure pathways for the purpose of setting Eco-SSLs. OSWER Directive 92857-55. U.S. Environmental Protection Agency, Office of Solid Waste and Emergency Response, Washington, DC. November 2003.

U.S. EPA. 2004. Risk assessment guidance for Superfund. Volume I: Human health evaluation manual, Part E, supplemental guidance for dermal risk assessment. EPA/540/R/99/005. U.S. Environmental Protection Agency, Office of Superfund Remediation and Technology innovation, Washington, DC.

U.S. EPA. 2005. Integrated risk information system (IRIS). http://www.epa.gov/iris/search.htm. U.S. Environmental Protection Agency.

University of Michigan. 1997. *Sorex arcticus*: Arctic shrew. http://animaldiversity.ummz.umich.edu/accounts/sorex/s._arcticus$narrative.html. Accessed on November 19, 2002. Last updated on April 3, 1997. University of Michigan, Museum of Zoology, Animal Diversity Web.

URS Team. 1996. Baseline ecological survey report, Adak Island, Alaska. URS Consultants, Science Applications International, Corp., and Shannon & Wilson, Inc.

USFWS. 1964. Pesticide-wildlife studies, 1963: A review of Fish and Wildlife Service investigations during the calendar year. FWS Circular 199.

USFWS. 1984. The ecology of tundra ponds of the Arctic Coastal Plain: A community profile. FWS/OBS-83/25. U.S. Fish and Wildlife Service.

USGS. 1995. 102. Arctic foothills. http://mapping-ak.wr.usgs.gov/research/ecoreg/region_02.html. Accessed July 9, 2001. Last updated August 25, 1995. U.S. Geological Survey, National Mapping Information, EROS Data Center.

USGS. 2002. Mercury in the environment. Fact sheet 146-00. www.usgs.gov/themes/factsheet/146-00. Accessed January 12, 2005. Last updated September 26, 2002.

USGS. 2003. The natural dispersal of metals to the environment in the Wulik River–Ikalukrok Creek area, Western Brooks Range, Alaska. Fact Sheet 107-03. U.S. Geological Survey.

van Dobben, H.F., H. Wolterbeek, G.W.W. Wamelink, and C.J.F. Ter Braak. 2001. Relationship between epiphytic lichens, trace elements and gaseous atmospheric pollutants. Environ. Pollut. 112:163−169.

Verschuuren, H.G., R. Kroes, and E.M. den Tonkelaar. 1976. Toxicity of methylmercury chloride in rats. II. Reproduction study. Toxicology 6:97−106.

Vitt, D.H., J.E. Marsh, and R.B. Bovey. 1988. Mosses lichens and ferns of northwest North America. Lone Pine Publishing, Edmonton, Alberta, Canada. 296 pp.

Vos, J.G., H.L. van der Maas, A. Musch, and E. Ram. 1971. Toxicity of hexachlorobenzene in Japanese quail with special reference to porphyria, liver damage, reproduction, and tissue residues. Toxicol. Appl. Pharmacol. 18:944–957.

Walker, D.A. 1996. Disturbance and recovery of Arctic Alaskan vegetation. Chapter 3. In: Ecological Studies, Volume 120. J.F. Reynolds and J.D. Tenhunen (eds). Springer-Verlag, Berlin.

Walker, D.A. 2000. Hierarchical subdivision of Arctic tundra based on vegetation response to climate, parent material and topography. Global Change Biol. 6(Suppl. 1):19−34.

Walker, D.A., and K.R. Everett. 1987. Road dust and its environmental impact on Alaskan taiga and tundra. Arctic Alpine Res. 19(4):479–489.

Walker, M.D., D.A. Walker, and N.A. Auerbach. 1994. Plant communities of a tussock tundra landscape in the Brooks Range Foothills, Alaska. J. Vegetat. Sci. 5:843–866.

Walker, D.A., J.G. Bockheim, F.S. Chapin, III, W. Eugster, F.E. Nelson, and C.L. Ping. 2001. Calcium-rich tundra, wildlife, and the "Mammoth Steppe." Quart. Sci. Rev. 20:149–163.

Ward, D.L., and T.J. Olson. 1980. Baseline aquatic investigations of fishes and heavy metal concentrations in the Kivalina and Wulik rivers, 1978–1979. GCO Minerals.

WDOE. 1995. Sediment management standards. Chapter 173-204 WAC. Washington State Department of Ecology.

WDOE. 1996. Model Toxics Control Act cleanup levels and risk calculations (CLARC II) update. Publication 94-145. Washington State Department of Ecology.

Weber-Scannell, P., and A.G. Ott. 2001. Aquatic biomonitoring at Red Dog Mine, 2000. Technical Report No. 01-04. Alaska Department of Fish and Game, Habitat and Restoration Division, Juneau, AK.

Webster, D.H., and W. Zibell. 2003. Iñupiat Eskimo dictionary. www.alaskool.org/language/dictionaries/inupiaq/default.htm. Accessed on January 23, 2003. Last updated 1998–2000. University of Alaska Anchorage, Institute of Social and Economic Research, Alaskool, Anchorage, AK.

West, G.C., R.B. Weeden, L. Irving, and L.J. Peyton. 1970. Geographic variation in body size and weight of willow ptarmigan. Arctic 23:240–253.

Wester, R.C., H.I. Maibach, L. Sedik, J. Melendres, S. DiZio, and M. Wade. 1992. *In vitro* percutaneous absorption of cadmium from water and soil into human skin. Fundam. Appl. Toxicol. 19:1–5.

Wester, R.C., H.I. Maibach, L. Sedik, J. Melendres, and M. Wade. 1993. *In vivo* and *in vitro* percutaneous absorption and skin decontamination of arsenic from water and soil. Fundam. Appl. Toxicol. 20:336–340.

Whitaker, Jr., J.O. (ed). 1997. National Audubon Society field guide to North American mammals. Alfred A. Knopf, New York, NY. 937 pp.

White, D.H., and M.P. Dieter. 1978. Effects of dietary vanadium in mallard ducks. J. Toxicol. Environ. Health 4:43–50.

White, D.H., and M.T. Finley. 1978. Uptake and retention of dietary cadmium in mallard ducks. Environ. Res. 17:53–59.

WHO. 1990. Environmental health criteria 107: Barium. U.N. Environment Programme, International Labour Organisation, World Health Organization.

Williamson, F.S.L., M.C. Thompson, and J.Q. Hines. 1966. Avifaunal investigations. pp. 437–480. In: Environment of the Cape Thompson Region, Alaska. N.J. Wilimovsky and J.N. Wolfe (eds). U.S. Atomic Energy Commission, Division of Technical Information, Oak Ridge, TN.

Wilson, D.E., and S. Ruff. 1999. The Smithsonian book of North American mammals. American Society of Mammalogists, Smithsonian Institution. UBC Press.

Witschi, J.C. 1990. Short-term dietary recall and recording methods. p. 52–68. In: Nutritional Epidemiology: Monographs in Epidemiology and Biostatistics. W. Willett (ed). Oxford University Press, New York, NY.

Xu, Y., J.P. Lay, and F. Korte. 1988. Fate and effects of xanthates in laboratory freshwater systems. Bull. Environ. Contam. Toxicol. 41:683–689.

Yadrick, M.K., M.A. Kenney, and E.A. Winterfeldt. 1989. Iron, copper, and zinc status: Response to supplementation with zinc or zinc and iron in adult females. Am. J. Clin. Nutr. 49:145–150.

YDRR. 2002. Yukon mammal series: Shrews. www.renres.gov.yk.ca/wildlife/shrews.html. Accessed on November 19, 2002. Yukon Department of Renewable Resources.

Yudin, B.S. 1962. Ecology of shrews (genus *Sorex*) in western Siberia. Akademiya Nauk S.S.S.R., Siberian Section, Trudy, Institut Biologii 8:33–34. (not seen, as cited in Aitchison 1987 and Buckner 1964)