DEC Review of Responses to National Park Service Comments (dated 11 July 2005) on the April 2005 Draft DMTS Fugitive Dust Risk Assessment

No.	Comment	Priority	Recommendation	Response	DEC Remarks
	Summary of Primary Concerns				
NPS-1	Reference areas do not represent true background conditions, but are located in areas contaminated with both fugitive dust and background mineralization. Reference areas are supposed to form the basis for comparison against more polluted areas. Findings of no significant differences between bioeffects in background versus polluted areas are therefore inappropriate. As used in the RA, this comparison has introduced bias.	High	Please review all background data provided by the NPS and determine if using lower background concentrations would affect the number of COPCs selected and conclusions about differences in site-versus-background risks. Summarize the findings of this analysis in the revised RA.	Additional figures and discussion of the NPS/Hasselbach data have been added in Section 1 describing nature and extent of fugitive dust deposition. Section 1.1 (Site Overview) is appended below, and the revised or added text is highlighted: *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. The mine area within the permit boundary (shown in Figure 1-5) is not addressed in this document. The areas from which data were collected and evaluated within the risk assessment are depicted in Figure 1-7.* *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 2003b, 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 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 2003b; Teck Cominco 2003). Sampling programs designed to support the risk assessment were conducted in 2003 and 2004 to obtain data*	Response is acceptable.

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				for additional analytes in multiple environments and media. These programs are described in the field sampling plans (Exponent 2003e, 2004a), and in Appendices A and E of 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 concentrate storage buildings 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 conceptual site model, is provided in Section 2.2.4. Teck Cominco continues to work on additional dust control improvements on an ongoing basis.* --- The uncertainty assessment for the human health risk assessment in Section 5.4.3 has been updated by adding the following sections of text: ***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).*	

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				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 2003b, pers. com., 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 Cape Krusenstern National Monument (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.*	

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				5.4.3.1.2 CoPC Screening *Selection of CoPCs for the human health risk assessment 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 sediment quality standards)* *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 7.3), 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 eliminated by comparison with reference areas, and those changes are related to fugitive dust deposition, the impact will be detected by concomitant changes in the concentrations of CoPCs 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 vs. 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 vs. 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*	

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				detected in the tissue comprising more than 90% 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. -- *The uncertainty assessment for the ecological risk assessment in Section 6.6 has been updated with a more detailed discussion regarding the selection of the reference areas, uncertainties associated with the reference area data, and their use in the assessment (including implications for CoPC selection). The new Section 6.6.1 (Uncertainties Related to Reference Area Selection), is included below:* *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.* ***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.*	

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				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 2003b, pers. com., 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 [$0.02 < P < 0.05$], among the samples with similar concentrations greater than 16 miles north of the DMTS). 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 Cape Krusenstern National Monument (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 CK1 and CK2 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 toxicity reference values (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 south CAKR. However, because south CAKR is well beyond the potential influence of the DMTS, it just illustrates the overly conservative nature of the aluminum TRV.*	

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				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.* ***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.* ***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.* ***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.*	

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				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, as described in Section 7.1.* Section 6.6.3. (Uncertainties Related to CoPC Screening), cited above, has been revised also. The revised text follows: *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 than 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 ecological risk assessment 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,*	

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				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 2-fold higher than the corresponding reference exposures. The following chemicals were eliminated from further consideration in the baseline ecological risk assessment, although their hazard quotients exceeded 1.0, because exposures at the site were less than 2-fold higher than 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, since 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 ecological risk assessment, 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	

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				dioxide toxicity literature is based on studies that have evaluated lichen as bioindicators of air pollution, as 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% 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 due to 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 include 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 1989). In laboratory experiments, Grace (1980) 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 SO_2 levels ranging from 23 to 40 $\mu g/m^3$ in Indiana.* *Liblik and Pensa (2001) summarized critical levels for SO_2 to 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 SO_2 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 SO_2 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 results identified limited potential risks from several of the CoPCs carried through the risk assessment. 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. During development of the risk management plan (discussed in Section 7.3), 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 eliminated by comparison with reference areas, and those changes are related to fugitive dust deposition, the impact will be detected by concomitant changes in the concentrations of CoPCs included in future monitoring programs.*	

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NPS-2	Lichens are inadequately studied in the RA but the small amount of data collected indicate a substantial lichen decline adjacent to the DMTS road. Impacts due to zinc and sulfur—elements widely implicated in lichen decline in the published literature—have not been addressed. Lichens and bryophytes were not identified by species so no information on species-level impacts or community change was provided. In spite of the wealth of publications on the toxicity of metals to lichens, the assessment of risk to lichens is based on only two publications.	High	Please review the existing literature on the toxicological effects of metals on lichen in the revised ERA. If the lichen work were repeated with a greater level of taxonomic resolution and other modifications suggested by the NPS (see NPS-16), would the overall findings be greatly changed? If so, to what extent? Provide answers to these questions in the revised ERA. Including more detailed lichen studies in future monitoring work at the site should be discussed.	The plant community surveys conducted in support of this risk assessment evaluated trends in frequency and cover for total mosses and total lichens. At this level of resolution, the assessment was able to discern trends with distance from dust sources and differences between site and reference areas that suggested that fugitive dust is adversely affecting moss and lichen communities in the study area. Had extra resources been devoted to speciation of mosses and lichens in the field, the overall conclusions of the risk assessment for terrestrial plants would likely stand. However, as noted in this comment, species-level data would identify the most sensitive components of the moss and lichen communities (the species most at risk). This information could be useful for elucidating the cause of the observed effects (e.g., metals toxicity or physical effects) and for focusing any future monitoring program on appropriate endpoints. The need for future study of plant communities (including lichen and bryophyte species) will be evaluated during development of the risk management plan. The risk management plan will define what actions need to be taken based on the findings of the DMTS risk assessment. The risk characterization for terrestrial plants (Section 6.2.3) has been expanded to include a more detailed discussion of the toxicological effects of metals on lichens. A portion of Section 6.2.3 is included below, and any revised or new text is highlighted: *Many nonvascular plants are more sensitive to metals than higher plants are. 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*	Response is acceptable.

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				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 CK1). 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 CK2; attached to this document). 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 CK2). 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. Another reason that these thresholds may be conservative relative to the DMTS is that copper and zinc often are more-than-additive in toxicity. *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 due to the zinc, but secondarily due 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 4 times greater than 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 higher than 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 which 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 the diminishing competition from the 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 due to secreting oxalic acid, which forms insoluble metal oxalates or fungal strands that regulate the amounts of metals reaching the thallus from metal-*	

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				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. Thus, 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 As, Br, Ca, Cd, Ce, Co, Cr, Cs, Fe, Hg, K, La, Na, Ni, Sb, Sc, Se, Sm, Th, and Zn) 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 SO_2 and NO_2 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; please refer to Section 6.6.2 – Uncertainties Related to CoPC Screening for review). 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.* In the conclusions section of Section 6.2.3, the following sentence was added: *Further study would be required to verify the lichen results and to define the nature and extent of lichen effects related to fugitive dust deposition from the DMTS port, road, and Red Dog Mine.* An additional literature review summarizing the toxicity of different sulfur species to lichens is provided in Section 6.6.3 (Uncertainties Related to CoPC Screening).The following text was added to Section 6.6.3 (Uncertainties Related to CoPC Screening): *Sulfur was eliminated from the list of CoPCs for the ecological risk assessment, 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*	

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				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, as 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% of the absorbed sulfur dioxide can be oxidized to sulfate and leached from lichens, which acts as a detoxifying mechanism. However, the retained sulfur dioxide can be converted to bisulfite, and can be toxic when accumulated at high levels due to acidification and necrosis of plant tissue. 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 include 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 1989). In laboratory experiments, Grace (1980 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 SO_2 levels ranging from 23 to 40 $\mu g/m^3$ in Indiana.* *Liblik and Pensa (2001) summarized critical levels for SO2 to 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 mentioned. Kashlina et al. (2003) found that the critical levels of SO_2 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 the Kola Peninsula of Arctic Russia, the critical level of SO_2 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.* *Kashulina, G., C. Reimann, and D. Banks. 2003. Sulphur in the Arctic environment (23): Environmental impact. Environ. Pollut. 124:151–171.* *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.*	

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				McCune, B. 1988. Lichen communities along O_3 and SO_2 gradients in Indianapolis. Bryologist 91:224–228. *Nash, III, T.H., and C. Gries. 2002. Lichens as bioindicators of sulfur dioxide. Symbiosis 33:1–21.* *Richardson, D.H.S., and E. Nieboer. 1983. Ecophysiological responses of lichens to sulphur dioxide. J. Hattori Bot. Lab. No. 54:331–351.* *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.*	
NPS-3	The RA fails to incorporate the spatial data of Hasselbach et al. (2004) in designing the siting of reference areas in areas known to be free of fugitive dust, or in analyzing the data beyond 1000 m from the DMTS road. These data could have formed the basis for analysis of impacts to lichen communities.	High	See recommendation for comment NPS-1.	See response to NPS-1.	Response is acceptable.
NPS-4	Muskox—a locally significant species with a small home range (unlike caribou)—were omitted from the RA. They consume large quantities of nonvascular plants, which uptake high concentrations of heavy metals relative to vascular plants.	High	Please discuss the fact that muskox are resident in the area. Please include a discussion of their habitat and feeding behavior. Please provide a rationale why the caribou is a more conservative receptor than the muskox. Please provide exposure parameters for review by Alaska DEC before completing the analysis.	Section 2.4.1 of the BERA (Site Description) noted that muskox sightings are frequent at the foothills of the DeLong Mountains and along riparian areas of the DMTS road. The text in this section was modified to indicate that they are resident species. When receptors are selected for evaluation in food web exposure models, they serve not only to estimate risk to that particular species, but also as an indicator species, results for which can be used to assess the likelihood of adverse effects to ecologically-similar species (i.e., those of a similar trophic level with similar dietary preferences and foraging habits). This approach eliminates the need to assess every species separately. DEC guidance (1999) recommends this approach by specifying default indicator species for different receptor groups and geographic regions of Alaska. In this risk assessment, the caribou serves as an appropriate indicator species for muskox, as the diet of the caribou is modeled as consisting of 80 percent nonvascular plants. Furthermore, caribou exposure scenarios evaluated small areas (e.g., port or mine assessment unit), which would be comparable to the lower end of the home range size for muskox. For example, Jingfors (1984) reports a core area, or home range, of 330 square km for a muskox herd inhabiting the Sadlerochit River in northern Alaska. Also, in the Arctic National Wildlife Refuge, radio collared muskoxen used an average core area of 223 square km in the summer and 27 to 70 square km in the winter (Reynolds et al. 2002). These studies indicate that muskox home range is not extremely small, and is comparable in size to the assessment units evaluated in this risk assessment. Thus, based on similar assumptions about dietary composition and home range size, the caribou is an appropriate indicator species for muskox in this ERA, and conclusions regarding risk to caribou are protective of risk to muskox.	Response is acceptable.
NPS-5	The RA uses a regulatory framework, rather than an ecological one. As a result, additive effects of metal toxicity, effects on areas beyond 1000 m, and effects to ecosystem members not represented by benchmark species (e.g., lichens, mosses) are under-addressed.	Medium	To the extent possible, the ERA should be revised to address additive impacts (especially for wildlife), effects beyond 1000 m from the haul road, and possible effects to non-benchmark species. Please clearly identify for the reader the elements of the existing work that do consider additive impacts (i.e. vegetation surveys, benthic surveys, sediment toxicity tests)	In the wildlife uncertainty section (Section 6.6.5.), we provided a discussion of the reasons why a simplified approach of summing hazard quotients is inappropriate for estimating additive impacts on wildlife. The last paragraph of Section 6.6.5.4 was revised as follows:	Response is acceptable.

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				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. Peraza et al. 1998. Effects of micronutrients on metal toxicity. Environ Health Perspect. 106 Suppl 1:203-16. Results of vegetation community surveys, stream benthic invertebrate surveys, and toxicity tests provide an indication of the magnitude of effects from cumulative exposure to multiple metals in soil, sediment, or water, as well as other anthropogenic or natural stressors. However, as documented in the risk assessment report, it is difficult to distinguish among multiple effects and assign causality to any specific chemical or non-chemical stressor, as illustrated in the case of evaluation of effects to plant communities along the DMTS road. Results of the risk assessment indicate that for most receptors, the potential for adverse effects appears to be largely restricted to areas less than 1,000 m from the road. For example, 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 metals, particularly 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. Similarly, changes in vascular plant community characteristics occur primarily within 100 m of the DMTS. As noted in the risk assessment, there is an indication that effects to lichens extend beyond 1,000 m from the road, but the distance over which these effects may be occurring is unknown, and may need to be evaluated as a component of any planned monitoring studies. The need for future studies (e.g., for lichens and mosses) will be evaluated during development of the risk management plan.	

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NPS-6	In spite of the serious ecological effects to vegetation on over 128 km^2 of tundra on NPS land, the RA concludes that no corrective action is necessary by Teck Cominco. NPS and USGS recently observed large problems with concentrate-contaminated vehicles and fugitive dust along the DMTS system. We believe these problems need to be addressed in a meaningful way.	High	Please revise the ERA so that adverse impacts to lichens and other receptor groups are not downplayed. The revised ERA should indicate that adverse effect thresholds have been exceeded for several receptor groups and that action is needed to further reduce fugitive dust emissions.	Several parts of the document have been revised and clarified in response to comments from NPS and other reviewers. Please see Comment NPS-112 for the revisions to Sections 7.2, 7.3, and 8.2. Regarding the need for actions, the risk assessment indicated that further actions to address the findings of the risk assessment would be identified through the development of a risk management plan, which will "evaluate risk management options within the general categories of institutional controls, engineering controls, monitoring, and remediation/ restoration… [and] identify the most appropriate combination of actions for management of risk in the short-term, and over the long-term life of the mine."	Response is acceptable.
NPS-7	While chronic effects are well-addressed in the document, the acute toxicity that may occur during snowmelt, as 7-8 months of deposited metals are released in a few weeks, is not considered.	High	Please identify the lack of evaluation of acute effects in the uncertainty section of the ERA. Future monitoring work should include studies to evaluate possible acute impacts during snowmelt.	To address this comment, the following paragraph was added to the end of Section 6.6.5.1.3 (Time Use)*:* *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 7.3.*	Response is acceptable.
NPS-8	Vegetation sampling was inadequate due to lack of true background reference conditions, failure to cover a broad variety of landcover types, failure to assure an adequate number of sample units, failure to identify the majority of plant taxa to species and failure to use plant species (rather than derived or composite variables) as the main inputs to plant community analysis.	High	See recommendations for comments NPS-1 and NPS-2. How do the shortcomings mentioned in this comment affect the results and conclusions of the vegetation survey work? Describe the effects in the revised ERA. Include more detailed vegetation analysis of more landcover types in future monitoring studies.	Analyses of existing plant community survey data were revised and supplemented as described in the response to comment NPS-21 below. The reanalysis does not change the fundamental conclusions of the draft risk assessment that tundra plant communities are affected by fugitive dust; that effects are most pronounced near dust sources; that the potential for effects is greatest for nonvascular species (bryophytes and lichens); and that further study would be required to delineate the full nature and extent of vegetation effects. The need for future studies will be evaluated during development of the risk management plan. The revised sections of the risk assessment regarding this comment are provided in the response to comment NPS-21. Please see also the responses to comments NPS-1 and NPS-2.	Response is acceptable.
NPS-9	The RA considers the toxicity of single elements well, but fails to base the RA in the biological reality that species face the additive effects from a suite of elements at the same time.	High	See recommendation for comment NPS-5.	Please see the response to comment NPS-5.	Response is acceptable.
NPS-10	The transects begin sampling at 10 m from the DMTS road. They thereby omit from study the areas with the greatest levels of contaminants present along the corridor in CAKR—the 1-10 m zone.	High	Please indicate in the revised ERA the magnitude of risk underestimation that may have resulted by excluding samples from the 1-10 m zone. See also comment USGS-31, which indicates the wildlife may be attracted to the area next the road due to early snowmelt.	Sampling conducted in an earlier phase of the investigation in 2001 had included collection of moss at 3 m from the road. During that sampling event, the field crew had noted plant communities that close to the road were greatly degraded. This degradation is most likely due to physical stressors associated with the road, such as flooding and impoundment of run-off water, dust fall, and gravel spray, although metals in dusts could also contribute to those effects. Based on the 2001 field observations, and to maximize data collection during Phase II sampling, the decision was made to start plant community surveys at 10 m from the road where physical effects, although still occurring, were somewhat less in magnitude. As noted in the risk assessment, adverse effects were noted on plant communities at 10 m from the road. Although not quantified in the ERA, effects of similar or greater magnitude would be expected for plant communities less than 10 m from the road.	Response is acceptable.

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				In the case of the small mammal trapping grids, however, the grids were centered at 20 m from the road, and extended inward almost all the way to the toe of the road shoulder (about 2-3 m from the road surface). Therefore, for food web models evaluating carnivorous wildlife, maximally exposed prey were included in exposure scenarios and results do not underestimate risk. For small mammals, possible adverse effects were noted at 10 m from the road based on exceedance of TRVs for some metals. It is possible that the magnitude of hazard quotients would be greater if food items were collected nearer the road; however, it is unknown to what extent poor habitat quality within 10 m of the road would limit use of that area as foraging habitat for small mammals.	
NPS-11	Even though the DMTS crosses 32 km of National Park Service lands designed to protect the ecosystem in perpetuity, Exponent has used industrial rather than residential screening levels in the RA.	Medium	Please clarify which screening values were used for the ecological risk assessment. Please indicate the extent to which risks may be underestimated based on using industrial versus residential screening values.	Ecologically based screening benchmarks were used for CoPC screening in the ecological risk assessment (see Tables 3-19 through 3-27). The screening levels referred to in the comment are human-health-based screening levels. Conservative residential screening levels were used in the human health risk assessment for the entire site (Section 3.3.1.2, Table 3-14). Non-residential screening levels were presented (Table 3-14) only as a frame of reference because the use patterns and potential degree of exposure for the site would more likely be approximated by non-residential (i.e., industrial) exposure assumptions. However, the nonresidential/industrial screening levels were not used to screen out CoPCs.	Response is acceptable.
NPS-12	Effects do not need to apply to an entire population or species to be significant.	High	Unsupported claims regarding no impacts to wildlife populations should be omitted from the revised ERA. Please assure that the ERA does not downplay or dismiss possible adverse impacts to wildlife.	The document has been revised to remove conclusions that appear to dismiss possible population-level effects. Additionally, the Uncertainty Assessment (Section 6.6.5.6 – Population Level Uncertainty) text has been expanded with a discussion of challenges associated with determining the spatial scales at which populations should be considered with regard to risk evaluations conducted as part of this ERA. The expanded text from Section 6.6.5.6 is included below: *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 Western Arctic Caribou Herd) that moves over vast areas of western Alaska. As discussed above, 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, up to 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,*	Response is acceptable.

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				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 represent a population "sink" where local environmental factors, including CoPCs, do not permit reproduction to occur at the replacement rate, and immigration of migrants from other sup-populations results 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 (e.g., within a kilometer from the road, within Cape Krusenstern National Monument, etc.). Broad-scale population surveys would be required to determine whether impacts to populations are occurring over these larger spatial scales.	
NPS-13	Bone and bone marrow is the locus of Pb accumulation in most fauna, but was not discussed in this risk assessment.	High	The revised assessment should include a discussion of the importance of bone as a site of lead accumulation and possible need for follow-up sampling to properly evaluate risks.	For the purposes of the food web exposure evaluations conducted as part of this ERA, sampling of bone lead concentrations is irrelevant for assessing risk. For all wildlife receptors, risk is determined by comparing daily dietary intake rates of metals to dietary dose TRVs. Evaluation of bone lead concentrations would only be relevant if the ERA used tissue threshold concentrations to establish TRVs. As a side note, in the case of the arctic fox and snowy owl, its diet includes small mammals, which were analyzed for whole body tissue concentrations, including bone. Thus, the food web model for the fox incorporates consumption of bone by a mammalian carnivore.	Response is acceptable.
	Primary Areas of Concern (detailed description)				
NPS-14	**Location of Reference Sites**. We do not believe that the Reference Areas represent true background conditions, as is their purpose in this RA. In theory, Reference Areas should be designed to capture concentrations and bioeffects of unpolluted, unmodified natural areas. Comparing the highly polluted "Site" areas to a "somewhat polluted" reference site--rather than a clean reference site-may potentially have led to erroneous conclusions of reduced (or no) risk in the highly polluted areas for some ecological components. As currently designed, the two primary Reference Areas are located only approximately 2 miles south of the DMTS haul road or the Port Site. Both of these areas occur in areas of likely heavy metal deposition as per Hasselbach et al. (2004). Though the Terrestrial Reference Area lies outside the area mapped by Hasselbach et al. (2004), it is reasonable to predict that the isolines showing enrichment in mosses would continue similarly around the northeast part of the road and the mine. Indeed, heavy metal enrichment might even go farther from the center line of the road in this area as it is closer to the highly dust-enriched mine site.	High	See recommendation for comment NPS-1.	Additional figures and discussion of the NPS/Hasselbach data have been added in Section 1 describing nature and extent of fugitive dust deposition. The uncertainty assessment in Section 6.6 has been updated with additional discussion regarding the selection of the reference areas, uncertainties associated with the reference area data, and their use in the assessment. Please see response to Comment NPS-1 for the revised portions of both Section 1 and Section 6.6.	Response is acceptable.

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<tr><td></td><td>The main Terrestrial Reference Site is also likely to contain elevated concentrations of heavy metals because it is in close proximity (perhaps about 1 km based on Fig 1-4) to known metals deposits.

Table 3-4 shows that the Reference Area concentration of Pb in soils ranged from 9-142 mg/kg with a mean of 38.5 (Table 1 below). Concentrations on the low end of this range are common in non-mineralized areas, while the high end of this range and the mean occurs only in mineralized sites. Table 1 shows much lower mean values and ranges for both Pb and Cd in the entire CAKR area (Hasselbach et al. 2004) and in the background levels for the Arctic Contaminant Research Program (Jesse Ford, pers. comm.). Tests for significant differences with these data should be done before the final draft is issued.

Table 1. Comparison of Pb and Cd concentrations (mg/kg) in soils from the Terrestrial Reference Area in the RA, Hasselbach et al. (2004) and ACRP Arctic Alaska background.</td><td></td><td></td><td></td><td></td></tr>
<tr><td></td><td><table><tr><th>Study</th><th>Pb Range</th><th>Pb Mean</th><th>Cd Range</th><th>Cd Mean</th></tr><tr><td>Terrestrial Reference Area (Table 3-4 in RA)</td><td>9 – 142</td><td>38.5</td><td>0.2 -3.6</td><td>1.1</td></tr><tr><td>Hasselbach et al. 2004</td><td>8 – 83</td><td>18</td><td>0.07 – 0.75</td><td>0.27</td></tr><tr><td>Ford ACRP</td><td>3 – 22</td><td>10.8</td><td>0.05 - 1.7</td><td>0.46</td></tr></table>
The Pb and Cd soils data from Hasselbach et al. 2004 should have been used to identify clean reference sites. Exponent had acquired all relevant NPS data during the study design phase of the RA. No data is presented comparing the concentrations in the moss Hylocomium splendens of Reference Areas relative to Hasselbach et al (2004). This would have been the primary means to test whether the Reference Area was enriched with metals from fugitive dust. Three samples of Hylocomium appear to have been taken from the Reference Area (Appendix C-22) but they are not summarized and do not represent an adequate sample for statistical inference. For Pb the mean of these 3 samples was approximately 7.7 mg/kg. None reached the levels documented by Ford (1995: 0.6 mg/kg Pb) or Hasselbach et al. (2004: 1.1-2.0 mg/kg) in clean areas. From Table 3-19 it appears that the concentrations of Pb in tundra soil in the Reference Site ranged from 3-23 mg/kg. It is not known what portion of the concentrations in tundra soil result from natural plant uptake and decomposition</td><td></td><td></td><td></td><td></td></tr>
</table>

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	versus incorporation from airborne deposition, and we don't have analogous data from other studies. We can say, however, that these levels are higher than the levels in/on mosses in clean areas (<2 mg/kg, as per above.) NPS notes that the soils in CAKR are not heavily mineralized (Table 1 above), and are relatively free from natural heavy metal enrichment. In choosing the Reference Areas, we recognize that Exponent was trying to mimic conditions along the DMTS. By choosing a reference area with soil and dust-borne mineralization, however, the RA fails to adequately represent the clean natural below and above-ground conditions found on NPS lands. For NPS, this makes the comparison of polluted "Site" areas to theoretically (but not actually) clean "Reference Areas" all the more problematic. From the perspective of study design, we question the wisdom of using one single reference area to reflect the diversity of flora, fauna and wind/deposition patterns on the landscape. Clearly a landscape-level approach would have been far preferable as well as statistically more defensible.				
NPS-15	One additional point: reference stations for soil were sampled in material borrow sites for the road construction and maintenance. Surface portions of these exposed soils are undoubtedly enriched by fugitive dust and make poor reference locations.	High	In the revised RA, please describe sampling depths at the borrow sites and whether or not samples collected there are biased high. Describe the magnitude of the effect.	Samples from the borrow sites were collected from beneath the surface layer from borrow material that had not yet been excavated, as so as to avoid the effects of any dust deposition. The samples were composites of representative materials from several places within each borrow site. The following sentence was added to the end of Section 3.2.3.2 (Reference Soil): *The material site samples were composite samples collected from representative source material within each material site from beneath surface layers, where it had not yet been excavated or exposed to dust deposition.* In addition, Section 3.3.1.1 (Comparison of Site Soil Data with Reference Data) has also been updated with the information above (highlighted text includes the changes that were made): 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 it had not yet been excavated or exposed to dust deposition. 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.	Response is acceptable.
NPS-16	**Lichens**. NPS has strong concern about the finding that lichen cover along the DMTS haul road at distances even beyond 2000m is now only 20-50% of that in the	High	See recommendation for comment NPS-2. Please ensure that the revised ERA makes adequate use of existing lichen literature and does not downplay adverse impacts to lichens.	Please see the responses to comments NPS-2 and NPS-6. Certain issues raised in this comment cannot be addressed without additional data collection (e.g., speciation of mosses and lichens). These issues will be considered in	Response is acceptable.

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	Reference Areas (p. 6-33). From the perspective of park management, this suggests impacts on some unknown quantity of tundra greater than 128 km^2. Lacking a root system, nonvascular plants are highly adept at absorbing minerals from the atmosphere and water. As a result, nonvascular plants uptake far more heavy metals from airborne deposition than vascular plants in the same locale. Using the data in Appendices C and G-19, we calculated that the lichens analyzed for elemental composition contained 25 to 92 times the Pb and Cd concentrations of vascular plants in the same location. Because of their heightened uptake capacity for heavy metals, they are at increased risk for injury, mortality and physiological problems than vascular plants at the same levels of fugitive dust deposition. These plants represent a large portion of the vegetation in CAKR, and play important ecological roles including forage, N-fixation and shelter for invertebrates. Lichens are at the base of the winter food chain for caribou and muskox and adult caribou consume an average of 6 kg/day dw of these plants (Boertje 1984). The conclusion (p. 8-2) that the primary changes in vegetation community structure occurred within 100 m of the DMTS road and port appears curious in light of the findings above. Lichens are an integral part of the healthy tundra environment in Cape Krusenstern, and additional study is warranted to determine the extent of damage and appropriate corrective actions. The lichen component of the risk assessment included just two literature citations on metal toxicity to lichens and one additional study on road dust (Folkesson and Andersson 1988, Tyler 1989, Auerbach et al. 1997). Lichen are among the most sensitive members of their ecosystem and there is a rich literature on lichens as indicators of air pollution. Hundreds of papers worldwide (chronicled in the series "Literature on air pollution and lichens" in the *Lichenologist*) and dozens of review papers and books (e.g., Nash & Wirth 1988; Richardson 1992; Seaward 1993; Smith et al. 1993; van Dobben 1993) published during the last century have documented the close relationship between lichen communities and air pollution, especially metals, SO_2, and acidifying or fertilizing nitrogen or sulfur-based pollutants. Smelters on the Kola Peninsula in Russia have been responsible for widespread lichen decline in adjacent Scandinavia from both metals and SO_2 (Tommervik et al. 1998). This decline was large and severe enough to be detected via remote sensing imagery. Much of the sensitivity of epiphytic lichens to air quality apparently results from their lack of a cuticle and their reliance on atmospheric sources of nutrition.			the design of the risk management plan and any future monitoring plans. The following responses address specific concerns expressed in this comment: 1. We recognize the importance of lichen conservation in Cape Krusenstern National Monument to the National Park Service. The need for future studies (including species-level analysis of mosses and lichens, and monitoring) will be evaluated during development of the risk management plan. 2. Studies cited in this comment demonstrate zinc toxicity to lichens. The risk characterization for terrestrial plants (Section 6.2.3) has been expanded to include a more detailed discussion of the toxicological effects of metals on lichens. Please refer to the response for comment NPS-2 for the revised text for Section 6.2.3. Additional data collection would be required to relate lichen cover to zinc concentrations in moss from Hasselbach et al. (2004). Lichen cover would need to be measured at locations collocated with moss sampling stations from Hasselbach et al. (2004); currently, zinc concentrations in moss are not available for most stations where plant community surveys were conducted, and far more concentration data have been collected than plant community data. However, zinc concentrations in lichen samples are compared against lichen toxicity thresholds from the literature to show where lichens are most likely to be adversely affected by zinc at the site and where adverse effects to lichens are less likely (Table CK2). This table will be added to the risk assessment document. 3. An additional literature review summarizing the toxicity of different sulfur species to lichens is provided in Section 6.6.3 (Uncertainties Related to CoPC Screening). Sulfur was eliminated from the list of CoPCs for the ecological risk assessment, 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. As the comment indicates, 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. Please refer to the response to comment NPS-2 for the revised Section 6.6.3.	

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	The entire lichen section of the study should be reworked with the following guidelines: 1. Lichens and bryophytes need to be identified to species, as this is the only way to determine which species are being most impacted by pollution and which may be responding only to the physical/hydrological effects of the DMTS and normal road dust. The use of frequency of lichens lumped as a group provides no meaningful data on the lichen species impacted by pollution. A capable lichenologist and bryologist needs to be employed in this endeavor as in similar projects where nonvasculars represent the frontline of decline in the wake of pollution (e.g., Athabasca Oil Sands, numerous smelter studies, sulfur and nitrogen emissions studies such as the USDA/Forest Inventory and Analysis Program oversees.) In terms of diversity, it is estimated from adjacent areas that lichens represent approximately 45% of the flora in CAKR; mosses and vascular plants probably represent approximately 30% each (Thomson 1984, Hulten 1968, Steere 1978, Neitlich and Hasselbach 1998). It is therefore estimated that 75% of the flora is represented by nonvascular plants, giving these plants a high priority for conservation. 2. Zinc is highly toxic to lichens. The literature is replete with references to lichen declines related to zinc both on a microscale (e.g., dead lichens underneath galvanized fences and hardware) and macroscale (e.g., smelters; Nash 1972, Nash 1975, Nash 1988, Buck et al. 1999, Folkeson 1984, Pilegaard 1994). The data provided in Hasselbach et al. (2004) provides a landscape view of zinc deposition. The zinc model should be studied for correlation to ground-based effects on lichen cover. 3. Sulfur (presumably present as sulfide and sulfate) represents approximately 20% and 32% of the Pb and Zn concentrates, respectively (Table 3-1). This element is not at all treated in the RA, which is a significant omission. Sulfur oxides (SO_x) are among the greatest toxins to lichens and some bryophytes, as these organisms uptake the pollutants directly without benefit of cuticle guard cells (Nash and Wirth 1988, McCune 1988). Sulfur effects need to be carefully considered for lichens, bryophytes and vascular vegetation. We consider this RA to be incomplete without these data. We recognize that sulfide—the form that probably accounts for the bulk of the sulfur—is a different form of sulfur than the SO_2 and SO_4^{-2} that have accounted for the bulk of the research on S effects on lichens. However, the most likely oxidation path of sulfide is to sulfate, and the effects of sulfides are little studied. The				

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	timeline and proportion of the S likely to change to forms injurious to lichens are currently unknown. While more papers have addressed the effects of SO_x including dissolved forms of sulfate on lichens, there are a number of studies isolating the harmful effects of SO_4^{-2} as well (e.g., Showman 1992, Marti 1983, Newberry 1974). As there are currently a great deal of unknowns and high potential risk, this topic requires considerably more study. Page 6-20 of the RA discusses plant communities with both dead and unhealthy lichens close to the DMTS road. An extreme version of this condition is noted along TT7, near the mine's ambient air boundary. We do not yet know the condition of lichen communities in CAKR, but given the cursory attention paid to lichens in the RA, we argue that this entire topic needs thorough quantitative evaluation. In a follow up study and monitoring, lichens need to be specifically targeted as receptor organisms.				
NPS-17	**Failure to incorporate Hasselbach et al. 2004 landscape-level spatial data**. As DEC is aware, NPS released work in 2004 detailing spatial patterns of heavy metal deposition in Cape Krusenstern National Monument (CAKR) and adjacent areas. One major shortcoming of the risk assessment was the failure to use these data to choose reference sites that represented truly uncontaminated areas. NPS data could also easily have been incorporated into the risk assessment to look at spatial patterns beyond the 1000m transect endpoints. Lichens—for which evidence of reduced cover are said to extend beyond 2000m (p 6-29)—could have been studied directly for effects using the landscape-level deposition values generated, thus taking the work to a new level. Instead, Exponent primarily limited the work to 1000m from the DMTS haul road.	High	See recommendation for comment NPS-1. To the extent possible, use the work of Hasselbach et al. (2004) to better quantify the extent of adverse impacts to lichens in the revised ERA.	See responses to comments NPS-1 and NPS-2. The sections that were revised to incorporate the work of Hasselbach are included in those responses and in the text.	Response is acceptable.
NPS-18	**Choice of Receptors**. We agree with the general conclusions that terrestrial taxa are at greatest risk, but argue that several key receptors were omitted from this study: 1. Muskoxen. Muskoxen from a small herd in the CAKR area are year-round residents of the DMTS corridor area. They are active grazers in the area as evidenced from a high density of pellets on Turtleback Mountain (Mile 7 of DMTS) in 2005. They may be particularly susceptible to heavy metal bioaccumulation because a large fraction of their diet comes from nonvascular plants. One study of muskoxen diet on the Seward Peninsula found that greater than 40% of these animals' diet came from mosses, with some additional portion coming from lichens (Ihl and Klein 2001). Muskoxen are a species of high concern for NPS and area residents.	High	Please discuss the fact that muskox are resident in the area. Please provide a rationale why the caribou is a more conservative receptor than the muskox. Please provide exposure parameters for review by Alaska DEC before completing the analysis. E & E believes that the selected avian receptors (snipe and Lapland longspur) are adequate surrogates for the Montane-Nesting shorebirds.	With regard to the use of caribou and not muskox in the wildlife exposure models please see the response to comment NPS-4. We concur with E&E's conclusion that the selected avian receptors are appropriate surrogates for montane-nesting shorebirds and that risk conclusions for these species (snipe, Lapland longspur) are protective of other species, including bristle-thighed curlew.	Response is acceptable.

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	2. Montane-Nesting Shorebirds. This group of birds nests on sparsely vegetated hilltops in the Northwest Arctic, then flies to wintering grounds in various Pacific Islands. One species, the globally rare Bristle-Thighed Curlew, occurs in the study area and is a species of high concern for the Audubon Society (Audubon 2005, Brown et al. 2001). Although these birds also spend a great deal of time elsewhere during the year, they forage intensively in contaminated areas near the DMTS during the summer months. Many birds in this group are known to forage right along the DMTS corridor (Bob Gill, pers. comm.).				
NPS-19	**Action Levels and Legal Context**. NPS does not agree with the conclusion that future actions to clean up heavy metal contamination are unnecessary and we look forward to a dialog on actions to reduce future contamination in the DMTS area. Reclamation is a requirement before Teck-Cominco and NANA vacate the easement in Cape Krusenstern National Monument. Exhibit B of the January 31, 1985 Land Exchange Agreement (*Terms and Conditions Governing Legislative Land Consolidation and Exchange Between NANA Regional Corporation, Inc., and the United States of America*, which was ratified in Public Law 99-96 that amended the Alaska Native Claims Settlement Act) includes section B. 4. Abandonment. This section specifies that NANA (or its operator Teck Cominco) must furnish a reclamation plan to NPS prior to abandoning the road. The plan would: – Prevent future interference with drainage – Mitigate soil erosion – Protect water quality, fish and wildlife and habitat, threatened and endangered species and cultural and paleontological resources – Examine costs or road surface scarification, methods and benefits of recontouring material sites and road prism, removal of culverts for fish streams and alternative revegetation techniques. NANA is required to conduct reclamation research during the life of the project. Furthermore and related to the fugitive dust issue, the Agreement states NANA and its assigns are required to implement dust control measures as required by ADEC and after consultation with NPS. While TC has been beginning work on a closure plan, NPS believes that since impacts have already been registered, corrective actions should not wait until the abandonment of the DMTS, at which point the contaminant levels and bioavailability could be significantly worse than at present.	High	The revised ERA should indicate that adverse effect thresholds have been exceeded for several receptor groups and that actions are needed to further reduce fugitive dust emissions. The other corrective actions mentioned in this comment also should be considered for inclusion in the risk management plan.	Language has been clarified in the revised risk assessment document regarding exceedance of adverse effects thresholds. Please refer to the response for Comment NPS-12; the response includes the revised section from Section 6.6.5.6 (Population Level Uncertainty). Also, in Section 6.5.4 (Risk Characterization for Wildlife), the revisions that have been made in response to this comment are highlighted below: *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 the toxicity assessment (Section 6.5.3): 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, 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 a hazard quotient 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 exposure in that case.* ---	Response is acceptable.

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				In addition, Section 6.5.4.1.1 (Willow Ptarmigan) was rewritten: *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 percent UCL on the mean concentrations of mercury at the port and zinc at the port and mine exceeded the NOAEL TRVs. However, hazard quotients were fairly low (1.2–1.4), and mean exposures did not exceed NOAEL TRVs (Table CK3). 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 percent UCL exposure), but did not exceed the LOAEL TRV (Table CK3). However, in the mine assessment unit, the mean and 95 percent 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 percent UCL on the mean concentrations, respectively). These results suggest that adverse effects from lead exposure near the port and mine are possible, particularly for the most exposed individuals in the population.* --- In addition, the second paragraph of Section 6.5.4.1.2 (Tundra Vole) was revised: *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 nears 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*	

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				(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 mine or port facilities. -- In addition, the following sentence was added to the first paragraph of Section 6.5.4.1.3 (Caribou): *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.* -- In Section 6.5.4.1.6 (Common Snipe), the second part of the following sentence in the middle of the second paragraph was revised: *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.* -- In Section 6.5.4.1.7 (Tundra Shrew), the following sentence was revised: *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.* --- In Section 6.5.4.2.3 (Moose), the last sentence of the section was revised: *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.* --- In Section 6.5.4.3.4 (Black-bellied plover), the last sentence was revised: *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.*	

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				--- Sub-sections of Section 6.7 (Interpretation of Ecological Significance) also include revisions (highlighted) in response to the comment. First, all paragraphs in Section 6.7.1 (Terrestrial Habitats), with the exception of the first paragraph, were revised. The revisions are indicated within the italicized text below: *Adverse effects to wildlife receptors from fugitive dust releases are expected to be minimal for most receptors. Locations and receptors where NOAEL and LOAEL hazard quotients, or only LOAEL hazard quotients, exceeded 1.0 are summarized in Tables JS5a and JS5b, respectively. Table JS6 summarizes the number of LOAEL hazard quotient exceedances per number of sites evaluated for each receptor.* *Herbivorous small mammals (i.e., tundra vole and tundra shrew) inhabiting tundra within 10-100 m of the DMTS road near the port facilities or near the mine's ambient air/solid waste boundary (i.e., along transects TT6 and TT7) showed incremental risk from exposure to barium, and aluminum. 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. Therefore, if adverse effects occur to 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 or barium.* *Regardless, possible effects on individuals in these areas, such as reduced growth (the endpoint for the aluminum TRVs) or increased mortality (the endpoint for the barium LOAEL TRV), are unlikely to translate into regional population-level effects given the limited area where adverse effects could occur, uncertainties related to the derivation of aluminum and barium TRVs, and extrapolation of individual-level responses to population endpoints, as discussed above in Section 6.6. In addition, aluminum and barium TRVs were derived from studies using much more soluble and bioavailable forms of barium and aluminum than those found at the site. Also, the barium endpoints for mammals based on rat studies using these more bioavailable forms (i.e., hypertension for the NOAEL, increased kidney masses and reduced ovarian masses for the LOAEL) are not conclusive as to their potential for effects on the populations. For aluminum, no effects have been found in avian studies, and in mammalian studies, the only effects endpoint was a reduction in weight gain of offspring in the second and third litters of second- and third-generation mice.* *Aluminum and barium are therefore not expected to be the risk drivers, as a result of the low solubility and low bioavailability of the forms present on the site. This was also illustrated in recent bioaccessibility testing work (Shock et al. 2007). The results of that research 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. In the risk assessment described throughout this document, the bioavailability of metals in soils was assumed to be 100 percent.*	

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				The food web model results for terrestrial herbivorous birds (i.e., ptarmigan) suggest that adverse effects (mortality or reproductive effects) from barium and lead exposures may occur in individuals foraging near the mine, and that adverse effects from lead are also possible in individuals foraging near the port, particularly for the most highly exposed individuals. These effects, if occurring, could result in population-level effects in areas near the port or mine. However, as stated above, the barium TRVs may overestimate toxicity of the relatively low solubility, low bioavailability forms of barium found on the site. Along the length of the road, the likelihood of adverse effects to herbivorous birds foraging in these areas is low, as 95 percent UCL on the mean exposures did not exceed NOAEL or LOAEL TRVs, except for exposure to barium, which exceeded the NOAEL TRV (hazard quotient of 1.7). Therefore, although risks cannot be considered negligible to ptarmigan inhabiting areas along the length of the road, it is unlikely that effects, if any, would result in a population-level effect in this area. *For caribou, there is a low likelihood that over-wintering individuals may experience adverse effects from aluminum exposure, as LOAEL-based hazard quotients ranged from 2.2 to 2.5 across the site, and were about 3-fold higher than comparable reference area hazard quotients. However, based on the low proportion of the total herd that could possibly over-winter near the mine site and the uncertainty associated with the aluminum TRV, it is very unlikely that any individual-level effects (e.g., reduced growth) would lead to population-level effects for the entire WACH. No adverse effects are predicted for the vast majority of caribou that only visit the site briefly during migrations. Food-web models also indicate that exposure to CoPCs are unlikely to result in population-level effects to other large-bodied mammalian herbivores (e.g., moose), avian invertivores (e.g., Lapland longspur), and avian and mammalian carnivores (e.g., snowy owl and Arctic fox).* *In summary, the potential for adverse effects to wildlife are most pronounced in the first 100 m adjacent to the road or facilities (Table JS5b) and in general are not expected to occur at any substantial distance from the road, port facilities or mine ambient air/solid waste boundary. However, lichen covers at 1,000-m and 2,000-m stations were significantly lower than reference covers, suggesting that lichen effects may still occur at these distances from the DMTS road corridor. Furthermore, 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 verify the lichen effects observed at distances greater than 100 m from dust sources, and possibly beyond 2,000 m, to define the nature and extent of these effects on lichens, and to distinguish the relative contributions of causative agents, such as metals and road dust or other factors on lichen toxicity. Overall, results of the ERA show that adverse effects to the terrestrial habitats and receptors are largely restricted to localized areas adjacent to the DMTS road, the port facility, and the mine ambient air/solid waste boundary, as summarized in Table JS7.* Section 6.7.2 (Freshwater Habitats) includes the following revisions: *In general, adverse ecological effects are not predicted in streams that cross the DMTS road, based on multiple lines of evidence. First, the evaluation of benthic macroinvertebrate drift assemblages indicated that the overall*	

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				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 from Aufeis Creek and Omikviorok River compared to streams near the mine, and no consistent evidence of a road effect on fish metals concentrations in these streams (Ott and Morris 2004). In Anxiety Ridge Creek, where cadmium and lead concentrations in juvenile Dolly Varden 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 the no-effects thresholds (Table CS1). 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. 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 models 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. 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. 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 6-23). 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 models indicate a very low likelihood*	

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				of adverse effects to survival, growth, or reproduction of herbivorous wildlife potentially foraging at these ponds. *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 above 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 site ponds suggests that they would be less likely to support the diversity of ecological receptors that the larger, more permanent ponds that occur in the tundra along the DMTS road would. Therefore, any adverse effects in these ephemeral ponds would have less ecological significance than if similar effects were to occur in ponds scattered across the tundra.* -- The absence of calculated action levels in the risk assessment document was not meant to imply that no future actions are needed. This language has also been clarified in Section 7.2 (Ecological Risk Based Action Levels). The last sentence of the section now reads as follows: *However, the potential use of ecologically based action levels will be evaluated further in development of the risk management plan, as described below.* Section 7.3 (Risk Management Plan) was revised – revisions include the highlighted portions: *A risk management plan will be developed to address the issues identified by the risk assessment. 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 for management of risk over the life of the mine.* *As described previously, human health based action levels were not calculated at this time because human health risks are not significantly elevated. However, some ecological risks were identified, as described in Section 7.2. Human health or ecologically based action levels could be used as one component of a risk management strategy, e.g., as a tool for risk management associated with monitoring and/or with Teck Cominco's voluntary cleanup program. The potential need for and use of action levels will be further evaluated in the process of developing the risk management plan. If numerical action levels are determined to be needed, they will be calculated and included in the plan.*	

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				Development of the plan will be a collaborative process involving DEC and other stakeholders throughout the process of identifying and evaluating options, and determining an agreed-upon course of action. -- The risk assessment document clearly stated the need to evaluate additional actions to address risks identified by the risk assessment. An appropriate mix of additional actions will be identified during development of the risk management plan, as stated in the risk assessment document. Teck Cominco has ongoing efforts to improve source controls and reduce emissions, through their Environmental Management Systems (ISO 14001) program, and has already taken actions to remove and recycle higher-concentration soils in DMTS facility areas. The requirements of the easement agreement will be considered in identifying additional action items during development of the risk management plan.	
NPS-20	**Action Levels And Fugitive Dust Control Measures**. As addressed in Section 2.2.4, Teck Cominco has made operational improvements that have reduced dust emissions in several areas. Still, there are significant problem areas with respect to fugitive dust, as observed on a recent visit on June 15-16, 2005. The primary issues remaining are: 1. *Truck Contamination*. While the truck washing station at the mine site is a modest start, we observed that trucks were still covered with dark grime (presumably a mixture of mud and concentrate) and $CaCl_2$ mud. The washing station did little to remove contaminants from the undercarriage. Moreover the washing station only operates during the short summer season. We observed concentrate on the fenders as well. The dust containment system at the unloading facility at the port site was very dusty and the air was filled with fine particulates in that facility despite air suction filters, stilling curtains, and vibrators to knock concentrate into hoppers. Presumably, suspended dust was redeposited on the trucks that then distributed the dust out on the tundra on the northbound trip. Overall, while the new trucks with hydraulic lids over the concentrate-bearing trailers certainly represent an upgrade, truck contamination has not been adequately addressed. 2. *Mine Site*. A great deal of fugitive dust comes from all aspects of work at the mine site. The transects closest to the mine clearly showed this pattern, as did field observation of dead lichens and stressed vegetation. While some attempts were made to control dust (e.g., traffic separation, water trucks around the buildings) there was a large amount of dust coming both from the open pit itself and the facilities. We did not observe any dust palliatives at the open pit, which contains highly enriched ores.	High	See recommendation for comment NPS-19.	Section 7 has been revised to indicate that the potential use of ecologically based action levels will be evaluated further in development of the risk management plan; please see response to comment NPS-19 for the text revisions in Section 7. The corrective actions recommended by NPS will be considered in the development of the Risk Management Plan. The following sentence from Section 7.3 (Risk Management Plan) was modified as follows: *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 for management of risk over the life of the mine.*	Response is acceptable.

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	3. *Port Site.* There are currently no facilities to decontaminate trucks at the port site. There were numerous holes in the CSB that are continuing to exchange air (and presumably dust) with the outside, though the doors were closed. 4. *DMTS Dust Control*. On our recent visit (June 15−16, 2005), we observed that $CaCl_2$ was applied very thickly to some portions of the haul road, but that the palliative was thin or absent over other portions of the road. Where a thick coating of the palliative was applied, a reduced dust trail from vehicles was observed. In other areas, the dust trails (as videoed by us on the DMTS near the airport) were quite large. Dust on roadside vegetation was still so thick that we were able to collect samples readily just by shaking leaves and twigs into sample bags adjacent to Turtleback Mountain (MP 7) in CAKR. We appreciate the steps that have been taken thus far, and would like to see a great deal more attention placed on efforts to bring this problem under control. 5. *Winter Dust Control*. We appreciate the challenges of controlling dust during the long winter, but it is extremely important due to the fact that dust is likely to travel much farther over the smooth, windpacked snow and frequently windy conditions. To date, even in light of the heightened awareness of the fugitive dust issues, voluntary cleanup does not appear to have remedied the situation. We would like to see some concrete measures stipulated in the Risk Management Plan. We do not agree with Exponent's conclusion that future actions are unnecessary. NANA and its assigns must return the DMTS to the NPS at the end of the easement period in an acceptable condition that meet ADEC and EPA standards for management of a public park unit. We are concerned that if heavy metals are allowed to accumulate and change over time to more bioavailable forms, we may face a difficult situation in the future. We think all reasonable and feasible measures to limit metals pollution in the now industrial--but eventually to become public--area should be undertaken sooner, not later.				
NPS-21	**Design Of Vegetation Sampling** (Section 6.2.1.1). The primary question posed is in this section is "How does distance from the DMTS haul road and/or port influence the composition of vegetation communities?" Exponent writes in Section 6.2.1.2 (Statistical Methods) that the "individual species data are highly variable; thus, average cover for vegetative types, or functional groups, was used in the analysis." Analytically, the analysis of	High	To the extent possible with existing information, please revise the analysis of the vegetation survey data to address the shortcomings described in this comment. Does changing the data analysis methods based on this comment affect the overall conclusions of the vegetation survey? Describe differences in conclusions in the revised ERA. Please include more detailed vegetation survey work of more landcover types in future monitoring studies to address these NPS concerns.	Analyses of existing plant community survey data were revised and supplemented as described below. The reanalysis does not change the fundamental conclusions of the draft risk assessment that tundra plant communities are affected by fugitive dust; that effects are most pronounced near dust sources; that the potential for effects is greatest for nonvascular species (bryophytes and lichens); and that further study would be required to delineate the full nature and extent of vegetation effects. The plant community study was not designed to characterize every single species within	Response is acceptable.

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	plant communities should be done using a primary species x plots matrix, rather than using such derived variables as evenness or diversity or functional groups as primary variables. The vegetation should be readdressed using these guidelines: 1. Address the primary question directly with community data, not derived or composite variables. Analyze the microplot data via ordination using NMS (nonmetric multidimensional scaling) and using distance to road as a secondary (or explanatory) variable. Then present the Pearson correlation coefficients with the primary and secondary matrix. Additionally, group the microplots by distance from road and conduct MRPP (multi-response permutation procedures) to see if there are community differences with distance from road. 2. Address the primary question additionally by controlling for vegetation. Much greater statistical power on the primary question of road/dust effects can be achieved by placing transects in homogeneous landcover types. This entire section should be redone using GIS-based dominant NPS landcover classes as stratifying variables and increasing the sample size to obtain sampling adequacy within each landcover class of interest. That is, these transects should be located only *within* homogeneous landcover blocks; they should not run across two or more landcover types. This would allow Exponent to pick up on changes within landcover types such as species shifts, elimination of species sensitive to heavy metals or sulfur, etc. This would be particularly germane to lichens, as they may or may not occur in dwarf shrub tundra depending on a variety of physical factors and pollution. 3. As noted above, lichens need to be identified to genus and mosses need to be identified at least to groups (e.g., Sphagnum, feather mosses, acrocarpus mosses). These determinations can be made easily in the field with a normal hand lens. It would be far easier to suggest causation for the observed decrease in lichen cover close to the road if the lichens and mosses were identified to genera/groups. For instance, lichens are naturally far less common in the wet habitats that promote Sphagnum growth. If Sphagnum predominated the microplots close to the road (rather than feather mosses which are generally more upland), then causation is more likely to be physical or pH-related. If as noted (p. 6-20) other mosses or no mosses dominated, and the habitat is one that typically supports high lichen cover (e.g. mesic to dry well-drained open			the plant communities, but rather to examine vegetation trends to determine whether effects exist and whether further investigation or risk management activity is warranted. Toward these ends, the assessment was sufficiently sensitive to identify the presence of effects on tundra plant communities from fugitive dust, thus indicating the need for further consideration of this assessment endpoint. Multiple lines of evidence were evaluated to reinforce these conclusions. Certain issues raised in this comment cannot be addressed without additional data collection (e.g., speciation of mosses and lichens). These issues will be considered in the development of the risk management plan and any future monitoring plans. 1. PCA was not based on species-level data because many species were not present at many locations. The composite vegetation variables that were used provide more continuous measures of change. NMDS analysis based on existing data (species-level percent cover data, and data for categories of vegetative litter and non-vegetated cover) has been added to the report. In both analyses, ordination was based on the vegetation measures of the community, and then the resulting dimensions or factors were evaluated with relation to distance and other environmental variables. The revised Table 6-9 presents Spearman rank non-parametric correlations between rotated PCA factors and distance from dust sources, and revised Figure 6-5 shows the rotated PCA figure. NMDS analyses are presented in Figure ME2 and Table ME1. These revised and new figures are included as an attachment to this document. Also, Section 6.2.1.3.6. (Principal Component Analysis) was deleted from the document and replaced with the Section 6.2.1.3.6 (Multivariate Ordination Analyses), below: ***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 eigenvalues 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*	

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	low shrub tundra) then a poverty of lichens is more likely to indicate chemical effects. 4. Conduct the study using a much larger number of plots that cover the entire spectrum of landcover types and cover each landcover class adequately. As it stands, we are not convinced that the vegetation survey is based on a high enough number of replicates. Assess the adequacy of sampling by using species-area curves, and convey the sampling adequacy to the readers.			*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 ME2 shows the first two axes of the NMDS results, and Table ME1 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 ME2). 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 6-13). 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 6-12). 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 6-13). 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 ME2). Table ME1 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 6-10 and 6-11). Although moss and lichen covers also*	

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				increase with distance from dust sources (Tables 6-10 an 6-11), 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 ME2). 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 6-11). 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 6-10 and 6-11).* *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 ME1 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*	

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				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 ME1). 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.* 2. Terrestrial transect locations were selected to achieve multiple objectives, including sampling a gradient of metals concentrations and targeting dominant plant communities representative of the diverse environments that are subject to dust deposition near the port, in the central portion of the road, and near the mine. Attempts were made to place transects in areas of roughly homogeneous vegetation type to minimize natural (intercommunity) variability; transects TT5, TT3, and TT8 were all located in the same vegetation type, tussock tundra, according to baseline maps (Dames & Moore 1983a) and field observations. Positioning a 1-km transect in homogenous vegetation near the mine's ambient air/solid waste boundary was challenging because of the large topographic relief, and as discussed in the risk characterization (Section 6.2.3.3), differences in environmental conditions (e.g., aspect, slope position, substrate) between hillslope stations confounded relationships between vegetation parameters and distance from dust sources. Coastal lagoon vegetation composition was also sensitive to transect placement relative to the shoreline, as discussed in Section 6.4.2.4 and illustrated in Figure 6-3. Appropriate reference station locations were not selected until toward the end of the field program, so as to match the reference station landcover types to those sampled at the site. The limitations of the plant community data sets due to small sample sizes are discussed in the uncertainty assessment (Section 6.6.4.1.4). Sample size requirements and landcover class designations will be considered in the design of future monitoring studies. 3. The need for future study of plant communities (including speciation of moss and lichen species, and development of site-specific effects thresholds) will be considered during development of the risk management plan. See also the response to comment NPS-16. 4. A new figure (Figure ME1) has been prepared that presents species-area curves for vascular plants in terrestrial and coastal lagoon plant communities. 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 cumulative data are plotted along the x-axis in the order in which the microplots were	

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				evaluated in the field. The figure is attached to this document, and is included in the revised risk assessment in Section 6.2.1.3.6 (Multivariate Ordination Analyses), which is appended above in bullet #1. The graph of the coastal plain community stations shows that ten microplots were sufficient to capture most vascular plant species. In fact, no new species were identified after the fifth microplot at station TT5-2000, after the sixth microplot at station TT51000, and after the eighth microplot at stations TT5-0100 and TT5-0010. Similarly, the species-area curves for stations along the tundra transects (TT3 and TT8) seemed to plateau, with few new species added after the fifth microplot. Coastal lagoon communities had low species richness compared to terrestrial plant communities, and most or all species were identified in the first few microplots. At hillslope stations the species-area curves suggest that ten microplots were not adequate to characterize this diverse community. The information gained from the species area curves will be considered in the design of future monitoring studies. It is evident from the discrepancies between species richness and area richness estimates (summarized in Table 6-14) that ten microplots did not always capture all the species present 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 but were not captured in microplots were forbs (e.g., lousewort and buttercup) at station TT5-0010, primarily shrubs (e.g., blueberry and Labrador tea) at station TT5-0100, and forbs, grasses, and willows (e.g., polar grass and bog willow) at stations TT30010, TT3-0100, TT8-0010, and TT8-0100. Again, this information will be considered in the design of future monitoring studies. Uncertainties associated with sample size are discussed in Section 6.6.4.1.4 of the Uncertainty Assessment.	
NPS-22	**Acute vs. Chronic Effects.** The RA never mentions the possible acute effects that may occur during melt-off. Fugitive dust is deposited on the snowpack for 7-8 months per year. It is then released in a matter of days or a few weeks. For sensitive organisms such as nonvascular plants, this may constitute an acutely toxic window that may create greater physiological harm than low level steady deposition.	High	See recommendation for comment NPS-7.	Please see the response to comment NPS-7.	Response is acceptable.
NPS-23	**Single Elements versus Additive Effects**. One of the large uncertainties in the RA is the question of multiple toxic stressors may contribute to injury or mortality beyond that caused by any single element. The heavy metals in fugitive dust occur jointly, not in isolation. There is little discussion of how the concert of metals may amplify the effects caused by any single metal.	Medium	See recommendation for comment NPS-5. Please add more discussion of additive effects to the uncertainty section of the revised ERA.	Please see the response to comment NPS-5.	Response is acceptable.

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NPS-24	**Transect Design.** Given that the greatest loads of contaminants are found immediately adjacent to the haul road (1-3 m; Hasselbach et al. 2004), we find it inappropriate for the RA's transects begin at the 10m mark (unless perhaps a pre-existing reclamation plan had already specified removal of all surface materials up to the 10 m mark). While the zone immediately adjacent to the road may not account for a great amount of land, it crosses a broad area and represents the most toxic area in CAKR. The omission of this zone misleadingly reduces the highest concentrations of heavy metals found in the study. An additional point: only a relatively small amount of the sampling occurred on NPS lands. As land managers affected by fugitive dust, we would appreciate the knowledge of how specifically the dust is affecting biota in our jurisdiction.	High	See recommendation for comment NPS-10. Include more sampling on NPS lands in future monitoring work.	Please see the response to comment NPS-10 with regard to the rationale for starting sampling transects at 10 m from the DMTS road and the possible implications for reported risk estimates. Regarding sampling within NPS-managed public lands: At the time the transect locations were selected, the patterns of dust deposition along the road were reasonably well understood, based on the prior work that had been done. Based on this understanding, terrestrial transect locations were selected to achieve multiple objectives, including sampling a gradient of metals concentrations and targeting dominant plant communities representative of the diverse environments that are subject to dust deposition near the port, in the central portion of the road, and near the mine. However, the commenter's point regarding the limited data available within NPS-managed public lands is noted, and the possible need for future studies within those areas will be considered during development of the risk management plan.	Response is acceptable.
NPS-25	**Use of Industrial Rather than Residential Screening Levels**. The RA assumes use of non-residential (industrial) screening levels for Pb along the entire DMTS road corridor and sample areas. (Industrial screening levels for Pb in soil are 1,000 mg/kg dw and residential screening levels are 400 mg/kg dw.) The document should clearly specify what clean-up levels it is using or rejecting and whether EPA or ADEC clean-up levels are being used. Even so, the RA reports 168 out of 479 sample sites found Pb exceeded industrial screening levels. All of these except one, however, were within the port ambient air boundary. If the screening level were the residential level, it's likely that most or all sample sites would exceed this screening level. The rationale for using the industrial level is that no one lives along this road corridor and a safety factor of 0.1 is used to protect people from exposure to Pb. Given NPS's mandate to preserve the flora and fauna in perpetuity, we consider the industrial screening levels inappropriate for areas within CAKR's boundaries.	High	See recommendation for comment NPS-11.	Section 3.3.1.2 of the risk assessment states, "Maximum surface soil concentrations from the road and port were compared with residential screening levels, as prescribed in DEC (2000)." In this last paragraph of this section, the sentence describing exceedances of residential screening levels was clarified as follows: *Maximum site soil concentrations of 10 chemicals (aluminum, antimony, arsenic, barium, cadmium, iron, lead, manganese, thallium, and zinc) exceeded residential risk-based screening levels.* Non-residential screening levels were not used to screen CoPCs. See also the response to comment NPS-11.	Response is acceptable.
NPS-26	**Bone and Bone Marrow**. The methodology used to document heavy metal contamination of arctic wildlife fails to focus attention on bone and bone marrow, where lead is most likely to be accumulate, causing physiological impacts and potentially passing into the human and wildlife food chains. Bone and marrow contaminant concentrations should be reported for representative wildlife species, including shrews, voles, ptarmigan, musk oxen, and caribou collected from within areas where actual data and modeling indicated high heavy metal deposition.	High	See recommendation for comment NPS-13.	Please see the response to comment NPS-13.	Response is acceptable.

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NPS-27	**Emphasis on Population-Level Effects**. The RA minimizes effects on herbivorous and insectivorous small mammals voles and shrews), and caribou, by suggesting that "localized effects on individuals' growth, survival and reproductive performance are unlikely to translate into population-level effects because of the small proportion of the total numbers affected". Lethal or sublethal effects to any subset of the wildlife population should not be lightly dismissed, especially when dealing heavy metals expected to persist in the environment for many years.	High	See recommendation for comment NPS-12.	Please see the response to comment NPS-12.	Response is acceptable.
	Responses to Specific Points				
NPS-28	**P. xxii-xxiii, Terrestrial Habitats.** For nonvascular plants, the negative effects of road dust may be compounded considerably by metals because, as noted, these plants get all of their nutrients from the atmosphere. Data is needed specifically on the effects of Cd, Zn, S compounds, and Pb on lichens and mosses as well as the additive effects of these compounds—including normal road dust. It is also necessary to differentiate between chronic effects of steady deposition versus the acute effects that may be experienced during melt-off, as 7-8 months of metals are released within a short time.	Medium	See recommendations for comments NPS-2, 7, and 16.	The following sentence was added to the first bullet under *Terrestrial Habitats*: *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.* Also, please see the responses to comments NPS-2, NPS-7, and NPS-16.	Response is acceptable.
NPS-29	**Pages xxii and xxiii, Terrestrial Habitats, Lichen Cover, Bullet 2.** The reference areas should have sufficient variability to define unaffected lichen cover and be far enough away from road fugitive dust impacts to measure background conditions. This is not the case. Effects to tundra lichens beyond 100 m from the physical and hydrological effects of the road are unlikely as is normal road dust (see below).	High	The revised ERA should describe how deficiencies in the study design (specifically those mentioned in this comment) affect the overall conclusions of the work. Causes of lichen decline along the road should be reevaluated.	Bullet 2 was revised as follows: *Differences between reference plant communities and plant communities beyond 1,000 to 2,000 m from the DMTS road, specifically the 2- to 4.5-fold decrease in lichen cover (Table 6-10 and 6-11), may be a result of fugitive dust deposition. Further study would be required to verify the lichen results and to define the nature and extent of lichen effects related to fugitive dust deposition from the DMTS port and road and Red Dog Mine.* Please see also the response to comment NPS-1.	Response is acceptable.
NPS-30	**Page xxiii, Terrestrial Habitats, Port Facility Areas, Paragraph 2.** If the field research does not document physical impacts on the ground from construction or report the damage to the tundra area during construction, then the conjecture should be left out. The summary should focus on the findings of the research. Future investigations on tundra hydrology may be included in another study.	High	Please remove the conjecture in this section of the report.	The reference to CSB construction has been removed from this section.	Response is acceptable.
NPS-31	**Page xxiii, Terrestrial Habitats, Small Mammals Bullet.** Voles and shrews within 100 m of the road show adverse effects, but the document concludes that no population level effects are likely. We disagree with this suggestion. Wouldn't small mammal populations along the road corridor and near the mine and port facility be adversely affected? The definition of "population level" used here needs to be clarified.	High	See recommendation for comment NPS-12.	Please see the response to comment NPS-12.	Response is acceptable.
NPS-32	**Page xxiii, Terrestrial Habitats, Birds Bullet.** Similar comment as above on population level effects. The RA states that "site-wide" effects on ptarmigan populations is	High	See recommendation for comment NPS-12.	Please see the response to comment NPS-12 with regard to the discussion of population level effects. Specifically for the ptarmigan, in response to comments from DEC, risk to this receptor has been re-evaluated on an	Response is acceptable.

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	very unlikely. What is the definition of site-wide? From the NPS standpoint the site of concern is the road corridor and the affected area in Cape Krusenstern National Monument as far as enrichment of heavy metals are detected. **Page xxiii, Terrestrial Habitats, Caribou Bullet.** Some idea of the size of the over-wintering herd would be helpful here.			assessment unit basis (see Section 6.5.3.1.1). Therefore, for the purposes of this risk evaluation, populations are considered as the animals within that assessment unit. For example, the port assessment unit would include all the ptarmigan that potentially forage within 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 Executive Summary text (fifth bullet of the Terrestrial Habitats subsection) was revised based on these changes as follows: • Adverse effects to herbivorous birds (e.g., ptarmigan) are possible in individuals foraging near the port and mine, particularly the most highly exposed individuals. These effects, if occurring, could result in population-level effects in these areas. The likelihood of adverse effects to herbivorous birds foraging in the central portion of the road is low, as 95 percent UCL on the mean exposures did not exceed NOAEL and or LOAEL TRVs. Therefore, although risks cannot be considered to be negligible to ptarmigan inhabiting the central portion of the road, it is unlikely that effects near the road, if any, would have a population-level effect in this area. The following highlighted text has been added to the caribou bullet in the Executive Summary to provide a description of the size of the over-wintering herd: • *In addition, it is very unlikely that any individual-level growth effects, if occurring, would lead to population-level effects because of the very small proportion (<0.02 percent) of the total Western Arctic Caribou Herd (estimated at 430,000 individuals) that could possibly over-winter near the mine site.* -- The revised ptarmigan assessment from Section 6.5.3.1.1 (Willow Ptarmigan) is included below: *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 percent 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 are summarized in Table CK3. 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 percent 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 percent 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 percent UCL on the mean exposure to mercury in the port assessment area slightly exceeded the NOAEL TRV (hazard quotient of 1.2). The 95 percent UCL on the mean exposures to zinc at the port and mine also exceeded the NOAEL TRV*	

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				(hazard quotients of 1.3 and 1.4, respectively). Hazard quotients for willow ptarmigan did not exceed 1.0 in the terrestrial reference area. -- Additionally, section 6.5.4.1.1 was also revised. Please refer to the revised text for this section provided in comment NPS-19, above.	
NPS-33	**Page xxii, Terrestrial Habitats, Terrestrial Wildlife Bullet.** Again the area of concern should be carefully articulated. We are not concerned with animal populations in the entirety of Northwestern Alaska, and this study did not measure those effects that widely. We are primarily concerned with effects to terrestrial wildlife in the study area along the DMTS. The results should focus on this; they should not seek to minimize the impacts by expanding the area or the size of the population under consideration. The authors need to carefully define the study area and areas of concern and stick with them. They may vary by resource, but should not extend to the entire northwestern part of Alaska.	High	See recommendation for comment NPS-12.	Please see the response to comment NPS-12.	Response is acceptable.
NPS-34	**P. 1-2, paragraph 4. Dust control.** As noted above, efforts to reduce fugitive dust have not generally been adequate with respect to truck decontamination, dust at the mine site, dust along the DMTS road, and truck contamination at the unloading facility at the port site. We suggest that other methods be employed to prevent deposition of ore concentration on outer truck surfaces, and to more thoroughly removed dust prior to leaving enclosed areas (e.g., compressed air/vacuum).	High	See recommendations for comments NPS-19 and 20.	Comment noted and referred to Teck Cominco. Also, the following sentence was added to the end of Section 1.1: *Teck Cominco continues to work on additional dust control improvements on an ongoing basis.*	Response is acceptable.
NPS-35	**P. 1-2, paragraph 2**. As noted, there is no mention of Hasselbach et al. (2004) or use of these extensive data in the RA. While Exponent cites some data from Hasselbach (2003), pers. comm., they clearly know about this document and they cite other documents published in 2004. In order to be more complete, RA should cite this work, use its data, and address the implications it raises.	High	Please cite Hasselbach et al. (2004) in the revised ERA, use its data, and address the implications it raises. See also recommendation for comment NPS-1.	Please see response to comment NPS-1.	Response is acceptable.
NPS-36	**Fig 2-2 and Table 6-1, p. 2-23.** It is unclear why inhalation is considered a secondary source of exposure for small mammals, foxes, etc. We observed a red fox napping immediately adjacent to the haul road (approximately 5 meters away). It has been already stated that particles > 1µm (i.e., 98% of the roadside dust) is incorporated into the GI tract after inhalation. Are these additional sources accounted for as a subset of ingestion—and if so, does inhalation only refer to particles <1µm?	Medium	In the revised ERA, please provide example calculations to demonstrate the relative importance of the inhalation pathway.	Inhaled versus swallowed doses are compared in the following text, which was added to the end of Section 6.6.5.1.1 (Body Masses and Intake Rate Parameters): *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 volatiles, that could be readily absorbed into the circulatory system from the lungs (U.S. EPA 1993).*	Response is acceptable.

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				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%, while in contrast, the combined diet and soil ingestion pathways contribute more than 99.9% 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 PBPK 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.* *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. 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.*	
NPS-37	**Fig 2-2**. Uptake via surface deposition is not listed as a direct effect for mammals or birds. Any mammals (like muskoxen) eating forage laden with a layer of road dust (e.g., dusty willow leaves) would be eating fugitive dust directly.	Medium	In the revised ERA, please clearly indicate whether or not external dust contamination was included in the analysis of wildlife foods.	The samples were analyzed unwashed. For further information, please see the response to Comment NPS-106.	Response is acceptable.
NPS-38	**Table 2-3 Relative importance of potential human exposure pathways.** No data are given for Pb. Either this data should be included or a footnote is needed to explain why it is not included.	Low	Please add a footnote to the table.	Lead risks are evaluated using separate models that do not predict a hazard index. Thus, they are not directly comparable to risks from other metals. A footnote has been added in the appropriate location in Table 2-3 (now renumbered as Table 2-5, attached).	Response is acceptable.
NPS-39	**Page 2-3, Paragraphs 2 and 3, Spill Data.** The incomplete reporting and recording of Pb and Zn concentrate spills before 1995 is disturbing. We wonder about the potentially large size of these spills as operating procedures and equipment were not as sophisticated as they have become in recent years. We recommend the contractor consult with the National Response Center records to determine spill records before 1995. These may predate the ADEC records if Cominco reported those as required by law.	High	Please summarize all available information on pre1995 spills in the revised RA.	Teck Cominco conducted a followup evaluation of all of the former concentrate spill sites, and has completed survey, sampling, cleanup (where needed), and closure of these sites, including reporting to DEC on those sites where additional action was taken. Table WH1 provides summary information about each of the truck spills, and Table WH2 lists the closeout dates of the re-evaluation of each spill site, and the specific documents containing the closeout information.	Response is acceptable.
NPS-40	**Road Dust** itself is not listed in the RA as a possible concern (e.g., PM 10, PM 2.5).	Medium	Please summarize available information on PM 10 and 2.5 in the revised RA. Based on this information, determine if dust itself should be considered a stressor. If so, revise the RA accordingly.	Soil particle size and the contribution of dust to human exposure are discussed in Section 2.3.3.1.2. Please refer to the response to comment NPS-36 that provides the updated text that discusses incidental inhalation for ecological receptors.	Response is acceptable.

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NPS-41	**Section 2.2.4. Fugitive Dust Control Measures**. NPS appreciates that some efforts have been made to control fugitive dust. As noted above in "Action Levels and Fugitive Dust Control Measures", the dust is still a problem.	High	See recommendations for comments NPS-19 and 20.	Please see response to comment NPS-20, which shows the updated text regarding current dust control measures. Observations noted in NPS-20, regarding problem areas with respect to fugitive dust control, were noted after the risk assessment was submitted, and although these are useful observations, this subject matter will be addressed further in the risk management plan when evaluating actions to be taken, such as additional engineering controls and monitoring. Appropriate monitoring options will be evaluated in the risk management plan to achieve monitoring objectives defined therein. To further address this comment and NPS-34, the following sentence was added to the end of Section 1.1: *Teck Cominco continues to work on additional dust control improvements on an ongoing basis.* In addition, the last sentence of the first paragraph in Section 2.4.4 was revised to state the following: *Efforts to reduce fugitive dust emissions are ongoing. A chronologic summary of dust control improvements is provided in Appendix L.* In addition, one sentence in the second paragraph of Section 2.4.4 was revised to state the following: *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 2005), and included in Appendix L of this document.*	Response is acceptable.
NPS-42	**Section 2.2.** Most of the sources here are identified as potential, rather than actual. For instance Section 2.2.3.: "Dust can be generated from drilling, blasting, …" rather than dust "is" generated. We have observed dust emanating from all of these sources during visits over the past 6 years.	Medium	In the revised RA, please describe these activities as actual sources of dust.	Language changes have been made as suggested.	Response is acceptable.
NPS-43	**Page 2-25. Measurement endpoints**. The preliminary measurement endpoints used to evaluate the attainment of assessment endpoints of structure and function of plant communities are the range of concentrations of CoPCs in soil. For nonvascular plants which lack roots, airborne deposition, rather than soil should have been used. Moreover, uptake rates from airborne dust for nonvasculars are not known, and vary by species. We do know, however, that vascular plants uptake approximately 1-4% of the heavy metals uptaken by nonvascular plants. This entire area needs thorough research, and possibly some original lab work to choose appropriate nonvascular species, to document their uptake rates, and to determine physiological effects.	Medium	If warranted, modify the list of assessment and measurement endpoints. In future monitoring work, consider including studies to evaluate metals uptake by nonvascular species.	Uncertainties associated with CoPC screening in the terrestrial environment are described in Section 6.6.2. This comment refers to preliminary measurement endpoints; however, refined measurement endpoints are presented in Section 6.1.5 and Table 6-1. As discussed in this section, the measurement endpoints used to evaluate the effects to assessment endpoints such as the structure and function of plant communities were focused on evaluation of community-level parameters for these endpoints. Thus, in the baseline risk assessment, plant community surveys were conducted to directly assess changes in vegetation structure. The need for future studies of nonvascular plants will be evaluated during development of the risk management plan.	Response is acceptable.
NPS-44	**Page 2-4, Section 2.2.1, Road:** This section and section 2.2.4 don't mention when new haul trucks with hydraulic lids replaced the older smaller trucks.	Low	Please provide this information in the revised RA.	The time frame (fall 2001) has been added to these sections, and a new appendix referenced from Section 2.2.4 provides a chronology of dust control improvements.	Response acceptable.

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NPS-45	**Page 2-9. 2.3.1.1 Land Ownership and Management, Paragraph 1, Last Sentence.** This section should indicate Public Law 99-96, which was passed in 1985, enacted the 100-year easement.	Low	Please make this point clear in the revised RA.	Changes have been made to the text.	Response acceptable.
NPS-46	**Page 2-11, 2.3.2 Potential Receptors:** There are a few small fly-in lodges along the Wulik River and tributaries to it that lie within the zone with heavy metals enrichment according to Hasselbach et al. (2004). Lodge operators working there during the summer fishing and fall hunting seasons may be at risk of exposure to CoPCs because they work in the field annually. These operators should be advised of the fugitive dust report and the RA.	Low	Please indicate the number and locations of these lodges in the revised RA. Are risks to receptors at the lodges covered by the evaluation of risks to subsistence receptors? If so, state this in the revised RA. If not, conduct the necessary analysis to define risks for this newly identified group.	As noted in Section 2.3.2, "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." Year-round subsistence users would have the potential for more frequent and prolonged exposure to the site than recreational users or seasonal workers in the recreational industry. Thus, the risk assessment is also protective of these potential receptors.	Response acceptable.
NPS-47	**Page 2-14, Section 2.3.3 Potential Exposure Pathways:** We think a representative marine mammal such as ugruk (bearded seal) or beluga whale should be added to the list of subsistence foods important in the area. Even though marine sediment levels of CoPCs are low, Pb half life in bones of mammals is up to 20 years. Therefore longer-lived marine mammals could accumulate heavy metals over a few years.	Medium	See recommendation for comment NPS-13.	The marine environment was screened out in the screening assessment portion of the RA. The possible need for future studies in the marine environment will be evaluated during development of the risk management plan.	Response acceptable.
NPS-48	**Page 2-14, Section 2.3.3.1 Worker and Subsistence Use in the Terrestrial Environment, Paragraph 2.** Were the "reference conditions" cited in Exponent (2002) in Reference Areas? If so, these are likely to have somewhat elevated metals concentrations compared to true background conditions. See critique in "Location of Reference Areas" above.	Medium	Please provide the information needed to answer this question in the revised RA. If the subject reference data are biased high due to fugitive dust contamination, please indicate the magnitude of the effect and whether or not it affects the conclusions of the analysis.	The reference berry data referred to in Exponent (2002a) are from samples collected near Noatak and Point Hope, not the RA reference areas. See also response to comment NPS-1. To clarify, the following sentence from the second paragraph of Section 2.3.3.1 was expanded as follows: *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).*	Response is acceptable.
NPS-49	**Page 3-1. Sulfate and Sulfur**. SO_4^{-2} and S are not included in the list of CoPC's in spite of potentially serious harm to the ecosystem. SO_4^{-2} and SO_2 have been implicated in large-scale and localized lichen declines in Europe, Asia and North America. AMAP cites the western Brooks Range as acutely vulnerable to the effects of acidification and S deposition (AMAP 1998 fig 9-25). Given that the concentrates are between 20-32% S (presumably present as sulfide and sulfate), it is likely that damage would occur to nonvascular plants as sulfides become oxidized to sulfates.	Medium	Please add sulfur as a COPC for lichens. Based on available literature and existing site data, provide a discussion in the revised ERA on the possible relationship between sulfur deposition and lichen decline at the site.	Please see the responses to comments NPS-2 and NPS-16. Also, the following sentence was added to the second paragraph of Section 3-1: *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).*	Response is acceptable.
NPS-50	**Table 3-2, Page 3-2, 3-3.** We are puzzled that data by Hasselbach et al. (2004) is not included in the RA. There were many opportunities to assess biota and choose sites relative to deposition beyond the 1000 m transect ends that were missed by not using this data set. It is stated that data gathered in 2001-2003 is not used as it does not represent the most recent deposition levels and predates some of the control measures. This dust, however, is still present in the environment, and will become increasingly bioavailable through weathering.	High	See recommendations for comments NPS-1 and 35.	Additional figures and discussion of the NPS/Hasselbach data have been added in Section 1 describing nature and extent of fugitive dust deposition. Please refer to NPS-1 for the changes that were made to Section 1 regarding the NPS/Hasselbach data. Regarding areas beyond 1,000 m, please also see response to comment NPS-5. Regarding bioavailability, the ERA makes the very conservative/protective assumption of 100% bioavailability for all metals. The human health risk	Response is acceptable.

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				assessment also assumes 100% bioavailability for all metals except lead, for which EPA default and site-specific values are both used. The paragraph below was added to the end of Section 6.6.4.1 (Uncertainties in Plant Community Surveys) to address the issue of weathering and plant communities: *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.*	
NPS-51	**Page 3-3, Data Usability, Paving and Removal.** Though areas with new pavement and recently removed soils would no longer represent exposure to humans and wildlife, these areas once represented great exposures. The document should specify when and where the pavement and removal activities took place.	Low	Please provide this information in the revised RA.	The document text referred to in this comment has been modified as follows: ***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).*	Response is acceptable.
NPS-52	**Table 3-4. Reference Areas Enriched.** The mean concentration of Pb in soil in the Terrestrial Reference Area was 38.5 mg/kg. In Hasselbach et al. (2004) the median and mean concentrations of Pb in soil were 15 and 18 mg/kg. The range of Pb values in soil in Hasselbach et al. (2004) were 8-84 mg/kg. In the Terrestrial Reference Area the range of Pb values in soil was 9-142 mg/kg. It is highly probable that the Terrestrial Reference Area was located in a zone of enriched mineralization, a suggestion supported by its proximity to a known mineral deposit in Fig 1-4. Surprisingly, no data is presented except in Appendix C-22 (unsummarized) on contaminant concentrations in Hylocomium splendens in the Reference Area. Only 3 samples are shown. For Pb the mean was approximately 7.7 mg/kg. Comparing Reference Area concentrations in moss with Hasselbach et al. (2004) would have been the primary means to test whether the Reference Area was enriched with metals from fugitive dust. Little inference can be drawn from the 3 values in the appendix other than to say that none reached the background levels documented by Ford (1995: 0.6 mg/kg Pb) or Hasselbach et al. (2004: 1.1-2.0 mg/kg) Table 3-5 additionally supports the idea that the Reference Area is enriched with metals from fugitive dust in that the mean tundra soil concentration of Pb is 9 mg/kg with a maximum of 23 mg/kg. Presumably,	High	See recommendations for comments NPS-1 and 35.	Please see response to comment NPS-1.	Response is acceptable.

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	enrichment in this soil layer would derive not directly from subsurface soil interchange but from fugitive dust and to a lesser extent plant uptake remains. For *Hylocomium splendens*, a large portion of CAKR south of the Tahinichok Mountains falls in the range of 0.5 -2 mg/kg Hasselbach et al. (2004), which begins to converge with Ford's (1995) median arctic Alaska baseline of 0.6 mg/kg.				
NPS-53	**Page 3-8, 3.2.8 Comparison of Site Data with Reference Data.** Because the reference site area is likely enriched with DMTS fugitive dust and natural mineralization, the statistical comparisons of site data with the reference site are likely in error if the data is supposed to be compared to background levels.	High	See recommendations for comments NPS-1 and 35.	Please see response to comment NPS-1, wherein the selection and suitability of reference areas is described, including discussion of the appropriateness of statistical comparison of site and reference area data.	Response is acceptable.
NPS-54	**Page 3-10, 3.3.1.1 Comparison of Site Soil Data with Reference Data.** Material sites used for road repair have most definitely been affected by fugitive dust in recent years because they are close to the road and within the zone of enrichment. The reference soil samples would only be valid for subsurface analysis where the samplers took care to avoid mixing with surface layers. Though we agree with the results of constituents that are likely elevated in Table 3-4, the comparisons would be even more evident with cleaner reference sites.	Medium	See recommendation for comment NPS-15.	The material site samples were composite samples collected from representative source material within each material site from beneath surface layers, where it had not yet been excavated or exposed to dust deposition. Language in Sections 3.2.3.2. and 3.3.1.1 has been clarified.	Response is acceptable.
NPS-55	**Page 3-20, 3-21. Benchmarks**. Tundra soil data were compared to ORNL toxicological benchmarks for effects on vascular plants. We need a detailed study of toxicological thresholds for nonvascular plants. Currently, as noted above, only 2 references are used for this.	Medium	In the revised ERA, please indicate that literature benchmarks for nonvascular species are highly limited. Consider conducting studies to identify threshold concentrations for nonvascular species as part of future monitoring studies at the site.	The following sentence was added to Section 3.1.5: *There are very few screening benchmarks available for nonvascular plants and therefore they were not used in the ecological screening assessment.* In addition, the following text was added to the uncertainty section (Section 6.6.2): *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."* In developing the risk management plan and associated future monitoring programs, existing and future data will be evaluated for possible use in development of site-specific effects thresholds for nonvascular plants.	Response is acceptable.

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NPS-56	**Page 3-13, Section 3.3.1.2, Last Paragraph. Human Health Screening Levels.** We note that Pb exceeded EPA non residential screening levels (1000 mg/kg) in 168 out of 479 samples and all but one of these sites occurred in road and facility areas within ambient air boundaries of the port. We would like to know how many Pb samples exceed the EPA residential screening level (400 mg/kg). The selection of health CoPCs for other metals assumed the residential exposure level. Since Pb is a major heavy metal of concern in the region, we wonder why it is treated differently.	Medium	Please clarify this section so that readers will not be confused regarding the screening value actually used for lead in soil. Please indicate where in the report the number of exceedances of the residential screening value (400 mg/kg) is found.	All metals were treated the same in the screening process. Only conservative residential screening levels were used to screen CoPCs (Section 3.3.1.2). Soil lead exceeded the residential screening level in 279 of 479 samples (Table 3-14). Thus, lead was retained as a site CoPC in the terrestrial environment (Section 3.3.1.3).	Response is acceptable.
NPS-57	**Page 3-17, 3.3.3.1.2 Marine Environment, Last Paragraph.** As noted for the terrestrial reference area, the marine reference area falls within a zone that is subject to fugitive dust enrichment according to extrapolation from Hasselbach et al. (2004). Our observations during and after loading of ore concentrate during 2004 indicated a layer of ore concentrate blanketing the barge deck beyond areas partially enclosed by tarpaulins. Despite ongoing deck cleanup efforts, we must assume that the area of deposition extended beyond the deck surface into the surrounding waters and that additional material had been lost during transport. The statistical results comparing the site samples with the reference sites are likely to be inaccurate. Some CoPCs may have been inappropriately excluded from consideration and the comparisons between site areas and truly clean areas may be weak. Furthermore, we would like to see the variability in Pb concentration values reported. If the high Pb values would require analysis when compared to the reference site, then we think Pb should be included to err on the side of conservatism and because the reference site is likely also contaminated.	High	In this section, please provide additional discussion regarding the importance of lead in the marine environment near the loading terminal and how it was evaluated in the baseline ERA. Refer to the 2004 sediment data as necessary to address the stated concerns.	The marine reference area is located approximately 3 miles to the south of the port. 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. As described in Sections 3.3.3.2.2, 3.3.3.3.2, and 4.3, all sediment CoPC concentrations (including lead) have been below all screening criteria in the sampling events conducted in the years since the port shiploader upgrades were completed. However, some level of ongoing monitoring is warranted. The appropriate frequency for future monitoring in the marine environment will be evaluated during development of the risk management plan. Please see also the response to comment NPS-1.	Response is acceptable.
NPS-58	**Page 3-19, 3.3.3.3.1 Lagoon Environment.** Again, the reference sites are in close proximity to the DMTS and are within the zone of likely enrichment, especially since they are near the port facility. Given this, it is impossible to know if the site concentrations of CoPC's exceeded true background levels.	High	In the revised RA, please indicate the degree of possible contamination of the subject reference areas and to what degree site-to-background comparisons may be affected by it.	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. Mean sediment concentrations (from the 2003 and 2004 sampling events) in the two lagoons 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. CoPCs were not identified in the lagoon environment for the human health risk assessment, nor were any complete exposure pathways present at the site. As discussed in Section 2.3.3.3, the lagoon environment near the DMTS was not evaluated further in the human health risk assessment 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.	Response is acceptable.

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				The lagoon environment was evaluated in the ecological risk assessment (see Sections 3.6.2.4 and 6.4). Note that the lagoon sediment benthic invertebrate toxicity testing (Section 6.4.1) does not rely on reference comparisons. No effects were observed in sediment samples from site lagoons. Please see also the response to comment NPS-1.	
NPS-59	**Page 4-3, 4.2.1 Terrestrial Assessment, First Full Paragraph.** We note the pH values change with distance from the DMTS road becoming more alkaline closer to the road with natural acidic conditions father out. This phenomenon needs explanation in the document.	Medium	Please provide additional explanation of this observed effect in the revised assessment.	Further discussion has been added to Section 4.2.1, also referencing the later discussion in Section 6.2. 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. In response to this comment, the revisions to the fifth paragraph of Section 4.2.1 are included below: *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 approximately a three-order-of-magnitude difference 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.*	Response is acceptable.
NPS-60	**Figure 4-11. High Contaminant Values Reported from CAKR**. Transect TT2 is in CAKR. It appears that tundra soils bear approximately 800+ mg/kg Pb close to the DMTS road. Lichens appear to have approximately 200 mg/kg Pb. These values corroborate data in Hasselbach et al. (2004), though that study is far more detailed. In terms of benchmark-based risk assessment, some taxa may tolerate these levels, but it is unclear whether this would be true for sensitive nonvascular plant taxa. Regardless, NPS believes these high values are incompatible with the NPS mandate to protect this park	High	See recommendations for comments NPS-2 and 16. Please include more detailed lichen studies within CAKR in future monitoring work at the site.	Comment noted. Please see the responses to comments NPS-2 and NPS-16. The need for future study of plant communities (including lichen and bryophyte species) will be evaluated during development of the risk management plan.	Response is acceptable.

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	unit unimpaired. The highest Pb concentration reported in lichens from the Pacific Northwest's USDA/Forest Service Lichen-Air Program was 127 mg/kg in a highly polluted section of the Columbia River Gorge (Geiser and Neitlich 2005). This area has lost all sensitive lichen taxa due to NO_x and metals pollution from Portland, OR.				
NPS-61	**Pages 5-2 and 5-3, 5.2.1.1 Exposure Point Concentrations for Environmental Media.** We note the report defines subsistence use areas for Kivalina and Noatak from Dames and Moore (1983), but with changes in technology and access (more reliable ATVs and snowmobiles), subsistence users travel farther and faster than they did in 1983 and subsistence use areas have likely changed in the last two decades. More recent use data would lead to more accurate area calculations and potential exposures.	Medium	In the revised RA, please discuss the extent to which subsistence use over a larger area would change the risk estimates for subsistence users. If warranted, consider collecting updated subsistence use data as part of future monitoring work at the site.	In response to the comment, Section 5.4.3.7 (Fractional Intake) was updated as follows: *The 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 (FI) 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 only occur 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, an FI based just on berry harvesting areas would be the same as the FI that was calculated based on the entire harvest use area. And 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 further away (and off-limits). Thus, it is reasonably likely that the FI, as calculated, overestimates risk from the site.* *For subsistence food animals with large home ranges (e.g., caribou and fish), FI 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 already integrate the animals' 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 than 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. Subsistence use over a larger area would reduce the FI related to the site because it would increase the denominator (i.e., the total area used for subsistence harvesting and hunting), without affecting the numerator (i.e., the portion of subsistence use area on the site) in the FI calculation. A lower FI would result in lower risk*	Response is acceptable.

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				estimates. Thus, the FI likely greatly overestimates the fraction of metals in these animals that is attributable to the site. In addition, the results of the caribou metals evaluation (Appendix H) 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 FI 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 is2.9. 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 FI 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.* ---	

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				In addition, at the request of DEC, risks were also calculated using an alternative caribou FI of 0.2. This value was calculated using the area reported to have cadmium levels elevated above background by Hasselbach et. al. (2005) as the site harvest area. The following text was added to the last paragraph of Section 5.2.2.2.3 (Subsistence Food): *An additional set of risk estimates was calculated using an alternative caribou FI of 0.2 because of the uncertainty surrounding the amount of impact site metals might have on caribou metals concentrations, and because of the unique role of caribou in diet and culture to 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.* The appropriate degree of future monitoring of subsistence foods will be evaluated during development of the risk management plan.	
NPS-62	**Page 5-3, 5.2.1.2.1 Data Used to Calculate Fish EPCs.** ADF&G collected Dolly Varden from the Wulik River from 1991 to 2003, but these fish are distant from the DMTS and would likely have low Pb concentrations. Fish should be collected and tested from closer to the DMTS (New Heart Creek, Aufeis Creek, Straight Creek, Omikviorok River, Tutak Creek, and Ikalukrok Creek).	High	In the revised RA, please describe the extent to which the existing fish data represent the worst-case situation along the haul road. If they do not, fish should be sampled from more contaminated streams in future monitoring work at the site.	The subsistence foods database indicates that Dolly Varden are the most substantial fish portion of the diet. Dolly Varden spend summers feeding in marine waters, then in fall enter the Wulik, Noatak, Kivalina, and other rivers, where they overwinter (but do not feed). While Dolly Varden do enter the streams crossing the DMTS (e.g., Anxiety Ridge Creek and Tutak Creek) to spawn, they spend very little time there, since the habitat is not suitable for overwintering. Instead, they migrate back out to the Wulik after spawning. Dolly Varden metals concentrations do not appear to differ significantly between fall (when the fish are returning from marine waters) and spring (after overwintering in the Wulik) sampling periods, suggesting a lack of impact from freshwater metals concentrations that may be higher than background levels they would encounter elsewhere. For example, in the five year period between 2001 and 2005, mean spring and fall muscle lead concentrations were 0.03 (range: ND-0.36) mg/kg and 0.02 (ND-0.19) mg/kg, respectively ($p=0.46$). Mean liver lead concentrations were 0.02 (ND-0.16) mg/kg and 0.03 (ND-0.23) mg/kg, respectively ($p=0.70$). Mean kidney lead concentrations were 0.03 (ND-0.20) mg/kg and 0.02 (ND-0.14) mg/kg, respectively ($p=0.34$). There are two additional reasons why tissue metals concentrations of Dolly Varden from the Wulik provide the best, most conservative basis on which to calculate risks for fish consumption: First, the majority of fish consumed would be harvested from the Wulik, Kivalina, and Noatak rivers, with the Wulik providing the largest resource. It is most appropriate to draw from the fish resource that is actually used by people when constructing a risk assessment or any other health evaluation. Of the three rivers, the Wulik is closest to the DMTS and receives drainage from streams that cross the DMTS, and thus it would be the most likely to show an impact from DMTS metals. Therefore, use of fish data from the Wulik is the more conservative choice. A study conducted for Maniilaq (Scannell 2005) showed little or no difference between metals concentrations in Dolly Varden from the Wulik and the Noatak. For example, aluminum concentrations from the Wulik River had a median aluminum concentration of 7.7 mg/kg (range of 4.1 to 12.1 mg/kg), and fish from the Noatak River had a median aluminum concentration of 5.3 mg/kg (ranging from below method reporting limit to 14.7 mg/kg). Fish collected from the Wulik River at Kivalina had a median concentration of	Response is acceptable.

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				1.11 mg/kg cadmium in kidneys and 0.15 mg/kg cadmium in livers in July 2004, and a median concentration of 0.48 mg/kg cadmium in kidneys and 0.205 mg/kg in livers in February 2005. Fish collected from the Noatak River had a median concentration of 0.73 mg/kg in kidneys and 0.185 mg/kg in livers in July 2004 and a median concentration of 0.467 mg/kg cadmium in kidneys and 0.126 mg/kg in livers in February 2005. Whole fish from the Wulik River had a median cadmium concentration of 0.21 mg/kg, and ranged from below method reporting limit to 0.03 mg/kg, and all Noatak whole fish concentrations were below method reporting limit. Dolly Varden collected from the Wulik River and Noatak River contained concentrations of chromium in all tissues that were below method reporting limit, with the exception of one fish from the Wulik River that had 1.2 mg/kg Cr in gill tissue. Copper concentrations in Wulik and Noatak River fish were similar, with median concentrations of 3.4 and 3.95 mg/kg, respectively, and maximum concentrations of 4.77 and 4.6 mg/kg, respectively. Concentrations of lead were at or below the MRL in liver, kidney, muscle and whole fish in all samples from Wulik and Noatak Rivers. Gill tissues in Wulik River fish had measurable, but low, concentrations of lead, with a maximum concentration of less than 1 mg/kg lead. Concentrations of mercury were at or less than the method reporting limit in all fish tissues from the Wulik and Noatak Rivers, and values that were slightly higher than the method reporting limit were not likely significantly different from the MRL. Median selenium concentrations for whole fish from Wulik and Noatak River were 1.3 and 1.4 mg/kg, respectively, and ranged from 1 mg/kg to 1.6 mg/kg, and 1 to 2.2 mg/kg, respectively. Median zinc concentrations for the Wulik and Noatak River fish were 47.8 (ranging from 41.4 to 60.5 mg/kg) and 54.15 mg/kg (ranging from 38.7 to 72 mg/kg), respectively (Scannell 2005). Furthermore, metals concentrations for Dolly Varden collected from the Wulik and Noatak Rivers were generally the same as or less than concentrations in other areas of Alaska. For example, median aluminum concentrations in 24 whole-body Arctic grayling sampled by the Alaska Department of Fish and Game from ponds in the vicinity of Fort Knox Mine (Last Chance Creek ponds) were reported at 20.5 mg/kg, with a range of 2.2 to 168 mg/kg. These values are considerably higher than median values reported for Wulik and Noatak River whole body analyses (Scannell 2005). Similarly, the U.S. Fish and Wildlife Service reported whole body Arctic char from Kanuti National Wildlife Refuge as containing 424 mg/kg aluminum, and northern pike and Arctic grayling with median concentrations of 6.2 mg/kg aluminum in livers (range 1.8 to 36.7) and median concentrations of 1.9 mg/kg aluminum in muscle (range 11.7 to 22.3 mg/kg) from Koyukuk National Wildlife Refuge. Liver aluminum concentrations in fish from the Kanuti National Wildlife Refuge are slightly higher than Wulik and Noatak fish, and muscle tissues from Kanuti are slightly lower. Eisler (1985) reported concentrations of 40 and 25 mg/kg in kidney and liver, respectively, as evidence of probable cadmium contamination, and residues of 800 mg/kg in kidney or more than 15 mg/kg in whole body tissues as potentially life threatening. As reported above, Wulik and Noatak fish had cadmium concentrations that were considerably lower than concentrations defined by Eisler (1985) as causing harm to fish. Similarly, Eisler (1985) reported that tissue levels in excess of 4 mg/kg chromium are evidence of chromium contamination. Chromium concentrations in fish from the Wulik and Noatak Rivers were well below this concentration. Maximum copper concentrations from Wulik and Noatak	

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				Rivers were considerably lower than maximum copper concentrations for freshwater fish determined from Eisler's (1985) nationwide study conducted in the United States. Similarly, according to Scannell (2005), lead concentrations in Wulik River fish were considerably below levels considered contaminated by Eisler (1985). Selenium concentrations in northern pike and Arctic grayling from the Koyukuk National Wildlife Refuge had muscle and liver concentrations of 1.1 mg/kg (range 0.49 to 2.3 mg/kg) and 4.5 mg/kg (range 11 to 6.4 mg/kg), respectively, similar to the Wulik and Noatak River fish. Concentrations of zinc from the Wulik and Noatak River fish are similar to zinc concentrations reported by Eisler (1985) for fish nationwide that are not considered contaminated. Finally, in the winter when Dolly Varden would be present, water column metals concentrations in the Wulik River are generally higher (although still low) as a result of freeze crystallization than they are in the streams crossing the DMTS, which freeze completely (Thompson 2006, pers. comm.). Thus, fish harvested from the Wulik River provide the most conservative, and as described above, most representative exposure concentration data. *Thompson, M. 2006. Personal communication (email to S. Shock, Exponent, dated April 24, 2006 regarding fish in creeks crossing the DMTS. Teck Cominco Alaska Incorporated, Anchorage, AK.* *Scannell. 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.*	
NPS-63	**Page 5-3, 5.2.1.2.2 Data Used to Calculate Caribou EPCs.** Caribou analysis should also include bone and bone marrow testing because Native people cook, boil and eat all parts of caribou including bone. Pb accumulates in bone and the Pb half-life in bone is up to 20 years in people and large mammals but only a few months in muscle, liver and kidneys (AMAP 1998, pages 393, 397, and 784).	High	See recommendation for comment NPS-13.	The following sentence was added after the first sentence of Section 5.4.3.10.1: *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.* The following information was added to the end of Section 5.4.3.10.1: *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*	Response is acceptable.

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				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 this study 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.* As discussed above in the quoted text, explicit incorporation of bone marrow data, if available, is unlikely to significantly affect the results of the analysis. However, consideration will be given to the possibility of sampling bone marrow as part of the next caribou sampling event.	
NPS-64	**Page 5-4, 5.2.1.2.3 Data Used to Calculate Ptarmigan EPCs.** Why weren't reference area ptarmigan tested to determine "background" or comparative CoPC levels and EPCs? We would like to know the lower levels of exposure farther away from the DMTS. Also, ptarmigan should be collected farther away from the DMTS to determine true background exposures to CoPCs in the region.	High	Please refer to Table G-28 (Analytical results for PHASE2 ptarmigan tissue [reference]) in this section. In future monitoring work, consider collecting additional ptarmigan samples further from the site.	Detailed discussion of the ptarmigan sampling and analysis and a comparison between site and reference ptarmigan is provided in Appendix H. Metals concentrations were analyzed in both site and reference ptarmigan. However, as with caribou, site ptarmigan metals concentrations were conservatively treated as if concentrations were entirely site related in the risk estimates. Monitoring of ptarmigan will be considered during development of the risk management plan.	Response is acceptable.
NPS-65	**Page 5-6, 5.2.1.2.7 Estimation of edible tissue weighted-average concentrations for caribou and ptarmigan.** Muscle weights for ptarmigan should also include the legs and back muscles, which would further increase the percent of muscle in ptarmigan EPC calculations. These parts and the heart are routinely eaten.	Low	Please verify if leg and back muscle also is eaten. If so, they should be included in the analysis.	To address this comment, the following paragraphs were added to the end of Section 5.2.1.2.7: *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 the information provided in ADPH (2001, 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. Thus, the estimates used to calculate tissue weighted-average metals*	Response is acceptable.

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				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.*	
NPS-66	**Page 5-6, 5.2.2.1 Lead Exposure.** We note Pb exposure is estimated using blood Pb levels, but Pb resides only a short time in blood and its half life is up to 20 years in bone of humans and wildlife (AMAP 1998, pages 393, 397, and 784). To be complete the RA needs to analyze Pb levels in bone of wildlife.	High	See recommendation for comment NPS-13.	In accordance with standard practice and EPA guidelines (U.S.EPA 1994, 1999), lead risks are expressed in terms of predicted blood lead levels. However, models used to predict blood lead include exposure from soil, water, air, and food, as well as background blood lead. The blood lead models also take into account the cycling of lead through different "compartments" of the body, including storage in bone. Ultimately, blood lead is the parameter of interest because prediction of health effects in people is based on association with blood lead levels. Regarding the need to measure lead levels in bones of wildlife to evaluate risk to those wildlife receptors, please see the response to comment NPS-13.	Response is acceptable.
NPS-67	**Page 5-7, 5.2.2.1 Lead Exposure.** This section indicates assumptions used in the model were EPA default assumptions except soil concentrations. The document should briefly describe the EPA assumptions and whether they are appropriate for Northwestern Alaska and how they are changed for some factors in this analysis. Secondly, we wonder about the accuracy of the fractional intake of soil for employees working at the port site and mine of 0.09.	Medium	In the revised RA, please provide a discussion of the EPA assumptions and whether they are appropriate for NW Alaska. Please describe the rationale for the FI of 0.09. If it cannot be defended, the parameter should be changed in consultation with Alaska DEC.	Additional description of EPA default assumptions has been added to the risk assessment. Per agreement with DEC, and as described in response to comment HH-14, fractional intake (FI) of soil was applied to soil concentration rather than soil ingestion rate in the IEUBK child lead model in the revised risk assessment. This approach is consistent with U.S. EPA (2003d) guidance and the last paragraph of Section 5.2.2.1 of the risk assessment has been modified to reflect the change, as follows: *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.*[1] *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 is available. Specifically, site-specific data for soil concentrations, gastrointestinal absorption for soil, drinking water concentration, and dietary intake are available and were used in the model. In addition, the soil lead EPC was multiplied by the fractional*	Response is acceptable.

[1] 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.

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				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 below in Section 5.2.2.2 on 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. The last paragraph of Section 5.2.2.1.1 has also been modified to reflect the change, as follows: *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.* The modification described above does not apply to the adult lead model (ALM). Unlike the IEUBK, the ALM is a linear model. Therefore, regardless of where in the equation FI is applied, the results are equivalent. As described in Section 5.2.3.1.5 of the risk assessment, it is assumed that 100 percent of soil intake occurs at the site (i.e., FI=1.0) for the portion of time that a worker/ subsistence user is working. This is the most health protective assumption possible for FI. An FI of 0.09 is applied only to the portion of time that a worker is off work and potentially engaged in subsistence activities. Calculation of FI and its application in the risk assessment was done in accordance with agreements with DEC, and is described both in response to DEC comment HH-18 and in the revised risk assessment in Section 5.2.3.1.5.	
NPS-68	**Page 5-7, 5.2.2.1.1 Soil Lead.** We note this section reports the mean soil Pb concentration in the port area is 1,225 mg/kg. This exceeds both the EPA and ADEC industrial clean-up levels. We think the RA should recommend these more highly contaminated areas are cleaned up immediately or capped (paved over) to reduce the potential tracking and transport of this contamination to adjacent areas.	High	See recommendation for comments NPS-19 and 20.	Soil removal and capping are two possible actions that will be considered during development of the risk management plan. EPA and DEC cleanup levels are based on generic exposure assumptions and applied either where site characteristics are consistent with the generic assumptions or where site-specific evaluations are otherwise deemed unnecessary. One aim of the risk assessment process is to develop alternative site-specific cleanup levels if risks are determined to be elevated. In fact, there are numerous examples where site-specific evaluations resulted in lead cleanup levels well above the default values for both industrial and residential sites (e.g., U.S. EPA 1998, U.S. EPA 1999). Based on the types of potential exposures at the site and other site-specific characteristics, human health risks were not determined to be elevated for the site. This indicates that site metals concentrations are not elevated above levels that pose a risk to human health. *U.S. EPA. 1998. Administrative order on consent for time critical removal action. ASARCO Incorporated, Sandy Smelters site, Sandy Utah. EPA Docket No. CERCLA-VIII-98-16. U.S. Environmental Protection Agency, Region VIII.*	Response is acceptable.

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				U.S. EPA. 1999. EPA Superfund record of decision: California Gulch, Operable unit 9 Leadville, CO. EPA ID: COD980717938. EPA/ROD/R08-99/055. U.S. Environmental Protection Agency.	
NPS-69	**Page 5-18 and 5-19, 5.2.3.1.1 Baseline Blood Lead Level.** We agree that the 1991 blood Pb levels are not representative of current conditions, nor would they have detected much Pb from the DMTS because the mine had only been operating for a few years by then and at lower production rates than at present. The 2004 blood level sampling at Kivalina and Noatak by ADPH was very limited and not comparable to the 1991 blood sampling to make any conclusions. We think ADPH failed to make a sufficient effort to sample a greater percentage of residents in these villages (only 10 people sampled in Kivalina and 48 people in Noatak and no children at either location).	Medium	Please add to the uncertainties the limitations of the ADPH study and discuss any other provisions that are available to assess blood lead levels.	Additional discussion has been added to the uncertainty section of the risk assessment to address limitations in the blood lead studies. In addition to biomonitoring as a means to evaluate blood lead levels in a population, the other most appropriate method to assess potential impacts of environmental lead concentrations on blood lead is through modeling. As such, EPA's Integrated Exposure Uptake/Biokinetic (IEUBK) child lead model and EPA's Adult Lead Model (ALM) were used to evaluate potential lead exposure in the risk assessment. The second paragraph of the uncertainty section in Section 5.4.3.4 of the risk assessment has been revised to address limitations in the blood lead studies, consistent with DEC comment HH-23, as follows: *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 Table 5-17, the ALM predicted geometric means of 1.9 µg/dL and 1.7 µg/dL for the 30 percent and 9.7 percent bioavailability scenarios, respectively. Blood cadmium levels were similarly low.* In addition, the last paragraph of the section prior to the numbered bullets was revised as follows: *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:*	Response is acceptable.
NPS-70	**Page 5-19, 5.2.3.1.2 Soil and Dust Ingestion Rate.** We think Exponent erred in not using the ADEC requested soil ingestion rate of 100 mg/day in the RA. Rather they decreased the soil ingestion rate to 50 mg/day while at work. If a worker is at the mine or port facility or driving a truck along the DMTS, their exposure to and ingestion rate of fugitive dust would likely be greater than the standard 50 mg/day and potentially higher than 100mg/day. Assuming the lower default ingestion rate for areas along the DMTS-where known soils levels for Pb are elevated far above ambient arctic conditions—should require substantial justification.	Medium	Please use 100 mg/day as the soil ingestion rate for workers.	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, 1997) 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 adult lead model 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, the risk assessment has been modified to assume a soil ingestion rate of 100 mg/day for both the subsistence and worker portions of the subsistence/worker scenario. The impact on the model results of using	Response is acceptable.

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				the default soil ingestion rate of 50 mg/day rather than 100 mg/day was already evaluated in the uncertainty assessment in Section 5.4.3.2.1 (Soil Ingestion Rate), and therefore no changes were made to the risk assessment in response to this comment. Section 5.4.3.2.1 of the uncertainty assessment is below: ***Soil Ingestion Rate*** *A soil ingestion rate during subsistence activities of 100 mg/day was used as an input to the ALM, as requested by DEC during work plan comment resolution. However, this value 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.*	
NPS-71	**Page 5-20, 5.2.3.1.4 Gastrointestinal Absorption Fraction of Lead from Soil.** This section states the absolute bioavailability of Pb in Red Dog ore for adults ranges from 2.7 percent to 5.4 percent, with an average of 3.9 percent. If the Risk Assessment purports to err on the side of overestimating exposures, then the absolute exposure of 5.4 % should be used in the fractional intake calculations, not the lower average value. The RA should evaluate the potential exposures of the most at-risk persons, not the average person.	Medium	In the revised RA, please present risk estimates for lead based also on the maximum bioavailability.	The risk assessment evaluates lead risks based on both the EPA default bioavailability values for soil (30 percent for children, 12 percent for adults) and the site-specific bioavailability for Red Dog ore. The default bioavailability, which is far above the range for Red Dog ore concentrate, provides the conservative estimate of risk. The site-specific bioavailability provides a more realistic estimate, so it is appropriate to use the best estimate for Red Dog ore concentrate, which is the average. Also, as summarized Table 5-7 of the risk assessment, bioavailability of Red Dog ore concentrate lead decreased with increasing lead concentrations. Site soil lead concentrations are closest to the highest concentration used in the NTP study of 100 mg/kg, which was associated with an absolute bioavailability of 2.7 percent for children. Thus, the trend in results from the NTP study suggests that even the average bioavailability from that study may overestimate actual bioavailability. In fact, based on the data from the NTP study, both the average value and the lowest value would be conservative because most soil lead concentrations at the site are higher than those used in the NTP study. Calculation of risks using two separate bioavailability values is meant to provide a conservative bracket around the potential risks associated with the site. As discussed above, the trend in the NTP study was for bioavailability to decrease as lead concentrations increase. Because site soil lead concentrations are at and above the highest lead concentration used in the NTP study, it follows that soil lead bioavailability would more likely be represented by the lowest absolute bioavailability value from the study of 2.7 percent, or lower. Thus, use of the average absolute bioavailability value from the NTP study of 3.9 percent is a conservative value to provide the lower end of the bracket of potential risks associated with the site. The EPA default value provides a very conservative estimate of potential risks at the upper end of the bracket. Use of a value anywhere in the middle of the two values used would not provide particularly useful information for risk	Response is acceptable.

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				managers, particularly a value such as 5.4 percent that has little relevance to site conditions. As discussed in response to DEC comment HH-21, uncertainties associated with use of results from the NTP study have been added to the revised risk assessment both in Section 5.2.2.1.2 and in the Uncertainty Assessment (Section 5.4.3). The following paragraph was added to the end of Section 5.2.2.1.2: 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 evaluating risks using both the IEUBK model default absolute bioavailability of 30 percent and the site-specific value of 9.7 percent.	
NPS-72	**Page 5-32, 5.4.2.1 Risk estimates for Lead.** Again, we think the default Pb bioavailability of 12 percent and 3.9 percent may be low due to averaging.	Medium	See recommendation for comment NPS-71.	See response to comment NPS-71.	Response is acceptable.
NPS-73	**Page 5-33, 5.4.3.1.1 Soil Ingestion Rate.** The ALM is designed to use averages, but averages leave out considerations of people most at risk to high levels of Pb exposures. Moreover, the EPA guidance for 50 mg/day incidental ingestion rate is probably reasonable for much of the US, but the DMTS is unusual with greatly elevated levels of Pb and other heavy metals in the soil and surface vegetation. For this reason we think the ADEC recommendation of 100 mg/day of soil ingestion rate for workers and subsistence users in the area is more reasonable. The ADEC recommended cleanup level of 50 mg/day ingestion rate would be the difference they want to see between the likely existing condition and the minimum level industry should clean up to.	Medium	See recommendation for comment NPS-70.	The ALM and IEUBK models are designed to use averages for input parameters because they both apply a geometric standard deviation to the resulting blood lead estimate that addresses the issues of variation in exposure patterns and variation in the resulting blood lead level given a specific exposure. The potential for more highly exposed individuals, or for higher blood lead response to the same exposure, is addressed because results are given as a probability distribution. The specific lead concentration in the soil being evaluated is irrelevant to the soil ingestion rate used in the model. The effect of varying soil lead concentrations on the model is addressed by the soil lead input parameter. Although the soil ingestion rate of 50 mg/day is consistent with expected exposure patterns for the worker scenario being evaluated, and consistent with DEC (2002), and U.S. EPA (1996c, 1997) guidance, at the request of DEC a soil ingestion rate of 100 mg/day was used for all adult scenarios.	Response is acceptable.
NPS-74	**Page 5-34, 5.4.3.1.2 Soil Lead EPC.** Again, we feel the assumptions regarding Pb bioavailability need to be evaluated carefully.	Medium	See recommendations for comments NPS-70 and 71.	See response to comment NPS-71.	Response is acceptable.
NPS-75	**Page 5-36, 5.4.3.3 Discussion of ADPH Blood Lead Surveys, Paragraphs 1 and 2.** Though ADPH succeeded in sampling a low percentage of the total populations of Kivalina and Noatak villages for blood Pb levels in 2004, we would like to know the ranges of blood Pb levels recorded in addition to the geometric means for each village.	Medium	Please provide the requested information in the revised RA.	The requested information has been added to the discussion of this study in the risk assessment. The information below has been added to the second paragraph of Section 5.4.3.4: *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*	Response is acceptable.

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				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 Table 5-17, the ALM predicted geometric means of 1.9 µg/dL and 1.7 µg/dL for the 30 percent and 9.7 percent bioavailability scenarios, respectively. Blood cadmium levels were similarly low.	
NPS-76	**Page 5-36 and 5-37, 5.4.3.3 Discussion of ADPH Blood Lead Surveys, Paragraphs 3.** We do not think the childbearing female population between ages 18 and 45 are necessarily the best to evaluate as a target population because women lose blood Pb burdens through menstruation and child birth. Older men who have and continue to hunt are likely to be more at risk of high blood Pb and bone Pb concentrations.	Medium	Please provide further support of your use of childbearing females in the revised RA.	Although the ALM estimates lead exposure in women of childbearing age, the individuals being evaluated are the hypothetical fetuses and newborns of those women. Susceptibility to the effects of lead is much greater during childhood and fetal development than at any time during adulthood.	Response is acceptable.
NPS-77	**Page 5-36 and 5-37, 5.4.3.3 Discussion of ADPH Blood Lead Surveys, Bullets 1 & 3.** We maintain that the RA did not always use conservative assumptions to ensure sensitive individuals are protected. See comments above. The data sets between 1991 and 2004 from Kivalina and Noatak are not comparable data sets in terms of percent of population sampled. The 2004 data set is inadequate. It is encouraging, however, to read that 32 of 33 individuals show lowered blood Pb level between the two sample years, however, those who volunteered may not be representative of the whole population. Blood tests are also not sensitive indicators of total tissue lead loads. Another question would be to test how blood Pb levels changed for those who were measured in 1991 and have since worked at the mine and also participate in subsistence activities.	Medium	See recommendations for comments NPS-70 and 71. Please consider including blood lead monitoring of residents in Kivalina and Noatak in future monitoring work at the site.	See responses to comments NPS-69, NPS-70, and NPS-71. Although the results of the ADPH surveys are presented and discussed in the uncertainty section, they do not enter into the risk assessment process. Rather, they are provided for comparison. Although there are clearly limitations in the ADPH studies, they are still actual measurements from the community, and the results are consistent with the results from the risk assessment. Blood lead testing is the most widely accepted and best-validated biomonitoring tool for assessing lead exposure in individuals and communities (CDC 2002). A risk assessment is one way to evaluate the potential for exposure to the community as a whole based on environmental conditions and people's habits and activities, but a risk assessment cannot provide information on individual exposures. It is appropriate for environmental assessments to be conducted for individuals with elevated blood levels in conjunction with a biomonitoring program. All community members have access to blood lead testing through Maniilaq. In the event that an individual is determined to have an elevated blood lead, Maniilaq could investigate the potential source of exposure for that individual using the appropriate CDC and public health protocols. *CDC. 2002. Managing Elevated Blood Lead Levels Among Young Children: Recommend-ations from the Advisory Committee on Childhood Lead Poisoning Prevention. Centers for Disease Control and Prevention, U.S. Department of Health and Human Services.*	Response is acceptable.
NPS-78	**Page 5-41, 5.4.3.7.3 Ptarmigan.** Paragraph 3 of this section contradicts the finding that Pb is elevated in ptarmigan tissues along the DMTS. Better wording could be something like: "Lead concentrations appear to be elevated in ptarmigan tissues, but levels of other CoPCs are low. Results from the RA indicate human health risks would not be greatly influenced from consumption of small amounts of ptarmigan."	Medium	Please revise the wording as indicated in the comment.	The text has been modified to reflect the commenter's concerns.	Response is acceptable.
NPS-79	**Page 5-41, 5.4.3.7.3 Ptarmigan, Last Paragraph.** This sentence appears to be a conclusion for the entire subsistence food investigation rather than simply for	High	Please omit the subject sentence from the revised RA.	The text has been modified in accordance with DEC recommendations based on DEC comment HH-27. The exact wording of the revision was agreed upon during the DEC comment resolution meeting convened by conference call on January 30, 2006.	Response is acceptable.

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	ptarmigan. Nevertheless, we believe a more qualified conclusion would be advisable.			The last three paragraphs of Section 5.4.3.7.3 have been revised as follows: *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.*	
NPS-80	**Table 6-9, Figures 6-5, 6-6. PCA Results**. Distance to DMTS road is highly correlated with Factor 2 (r^2=-0.48), as are a suite of heavy metals (0.4< r^2<0.8). PCA is not a preferred method of ordination as it is known to distort plant community data. NMS (nonmetric multidimensional scaling) has become the modern standard, and may lead to very different values. If lichen cover increases with distance to DMTS road, we wonder why this isn't showing up on Factor 2. Possibly the lichen signal is swamped out in the ordination by the major community differences related to physiography—or perhaps it owes to the fact that the ordination results are based only on vascular plants. To get a better sense of the problem for nonvascular plants, they need to be identified to species and included as part of the plot x species matrix that gets ordinated. They may also be ordinated by themselves, or in concert with bryophytes in a reduced matrix, for additional explanatory power. Equally, subsets of the main matrix may be ordinated by themselves (e.g., landcover classes known to be high in lichen cover, classes with greatest representation close to the DMTS road). NMS should be used for all ordinations. Heavy metal values from Hasselbach et al. (2004) should be used as environmental variables along with sulfur. The PCA results presented here are additionally misleading because only composite values (diversity, evenness, etc) are used in the primary ordination—rather than actual plant community data—and lichen cover is used as an explanatory variable rather than a member of the community. Moreover, all possible environmental variables need to be overlaid into the ordination as explanatory variables so that the axes may	High	Please reanalyze the existing plant survey data based on the recommendations in this comment. Please include more detailed vegetation survey work in future monitoring studies at the site. See also recommendations for comments NPS-16 and 21.	PCA was not based on species-level data, because many species are not present at many locations. Rather, composite vegetation variables were used to provide more continuous measures of change. As requested, NMDS analysis based on species-level data has been added to the report, as well as rotated PCA factors. Both ordination analyses were based on the vegetation measures of the community, including lichen and moss cover, and then the resulting dimensions or factors were evaluated with relation to distance and other environmental variables. Please see the response to comment NPS-21 for detailed discussion of the additional analysis conducted, including the revised text, tables, and figures that are associated with this part of the revised risk assessment. As presented in the tables for the response to NPS-21, the following elements were addressed by the revised analysis: antimony, arsenic, barium, cadmium, cobalt, copper, lead, manganese, mercury, molybdenum, selenium, silver, thallium, vanadium, and zinc. In the absence of sulfur data, sulfur was not included in the analysis.	Response is acceptable.

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	be interpretable. Axes need to then be rotated to achieve interpretability with the major explanatory variable of interest. The ordination approach presented in the RA would not survive most standard peer review in vegetation or ecological journals.				
NPS-81	**Appendix C-21. Some Very High Metals Levels In Nonvascular Plants**. Pb concentrations in Hylocomium are presented for PO-05m (1670 mg/kg) and TT1-0100 (Phase1RA). There is no mention of TT1 in the Risk Assessment. Where is this? Additionally, 1500 ppm of Pb is reported for the lichen Cladina sp. Values of this magnitude are typically accompanied with injury and/or mortality from multiple stressors.	High	Please include the data referred to in this comment in the baseline ERA. Please ensure that the revised ERA does not downplay adverse impacts to lichens.	Transect TT1 was positioned downwind of the CSBs. Station TT10100 was located approximately 100 m downwind from CSB1. Tundra soil and moss data were collected at this transect in 2003. An area of nearly complete plant mortality was observed at the northwest corner of CSB1; thus, this area was not suitable for assessing food source concentrations to evaluate receptor exposure, because it was not an area suitable for habitation or foraging by receptors. Therefore, this transect was not included in the following sampling program in 2004, wherein these types of samples were collected. Instead, transect TT5 was added nearby, extending from the DMTS road past the loop road and beyond, to 2000 m from the DMTS. However, moss zinc concentration data from stations TT1-0100 and PO-05M, as well as other stations, were compared to phytotoxicity thresholds in Table CK1, and data for all CoPCs were incorporated into the food web models for the caribou foraging in the port assessment unit. Moss concentration data are illustrated on Figure 1-9. The 1,500 ppm lead value in *Cladina* lichens was reported for a sample collected in a snow accumulation area at the edge of the mine's ambient air/solid waste boundary (station TT7-0010). Some signs of lichen decline were noted at station TT70010 (Section 6.2.3.1.2, Summary of Field Observations), but plant communities along TT7 were not assessed directly through community surveys. Field schedule limitations precluded full plant community assessments at this transect.	Response is acceptable.
NPS-82	**Page 6-6; Section 6.1.6. Muskox**. As noted above, we are disappointed that muskox were not chosen as a receptor. They consume large quantities of moss and lichens, which absorb 25 to 100 times the amount of metals as vascular plants. They also have a much smaller home range and their pellets are found in abundance along the DMTS.	High	See recommendation for comment NPS-4.	Please see the response to comment NPS-4 regarding the use of caribou in lieu of muskox in food web exposure models.	Response is acceptable.
NPS-83	**Table 6-3. Lichen distribution related to distance from DMTS road**. Lichen cover again emerges as significantly different ($p<0.05$) between the site and reference area and especially different (0.03) at the 10m distance. Again, sulfur forms should be added to this table. Table 6-4 amplifies the high correlation between distance to the DMTS road and lichen frequency and lichen cover ($r^2=0.77$).	Medium	See recommendations for comments NPS-2 and 16.	Comment noted. Please see the responses to comments NPS-2 and NPS-16.	Response is acceptable.
NPS-84	**Page 6-12, 6.2.1.1 Plant Survey Methods, Paragraph 2.** The vegetation communities along transect TT2 near the port's ambient boundary and TT7 downwind of the mine's ambient air/solid waste permit boundary, were assessed qualitatively without formal plant community characterization. We wonder why these sites were excluded. As highly polluted sites, these sites would have been most informative.	High	In future monitoring work at the site, please include more detailed evaluation of the vegetation along these two transects. See NPS recommendations in comment NPS-21.	Please see the response to comment NPS-21. The need for further study of plant communities will be considered during development of the risk management plan.	Response is acceptable.

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NPS-85	**Page 6-13, Plant Survey Methods, Last Paragraph.** To what extent would the date of sampling affect frequency and cover calculations? A brief discussion on this point would be helpful.	Medium	Please include a brief discussion of this issue in the revised ERA.	Uncertainties associated with the timing of the field event are discussed in Section 6.6.4.1.3 (Field Sampling Methods). Revisions to the text are shown below: *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.*	Response is acceptable.
NPS-86	**Page 6-20. Schematic layout for vegetation sampling: implications for contaminant loads**. In this analysis, graminoid communities are not different in the site versus reference area. Yet it is obvious from having conducted studies in the area that grasses flourish unnaturally immediately along the road corridor (1-3 m from the road). The cause of the grass bloom is probably the nutrient enrichment from road dust. This zone was omitted from the study. The omission of this 1-10 m zone also diminishes the potential contaminant loads found in the study tremendously.	High	To the extent possible with existing information, please describe impacts to vegetation in the 1-10 m zone in the revised ERA. In future monitoring work, please include vegetation survey work in the 1-10 m zone.	Please see the response to comment NPS-10. Vegetation observations at the 10-m stations are described in Section 6.2.1.3.2 (Summary of Field Observations) and illustrated in Photographs 33, 39, and 45. Vegetation effects observed at 10 m from the road are generally similar to effects within the first 10 m, except that the physical influence of the road is greater in the first few meters. For example, see Photograph 25, which shows impounded water and a stand of *Eriophorum angustifolium* adjacent to the road shoulder on transect TT2. Conditions were not surveyed everywhere along the road, so it is difficult to say how representative this photograph is of conditions adjacent to the road. It was perhaps one of the more obvious examples of graminoid occurrence next to the road. The commenter's observation that grasses flourish near the road is consistent with the results of road studies such as Auerbach et al. (1997), who found up to 2-fold higher graminoid biomass close to the Dalton Highway than at undisturbed plots. If the area within 1-3 m of the DMTS road were further surveyed, a difference would likely be observable between site and reference occurrence of graminoids. The reasons for the station spacing and layout are described in the response to comment NPS-10. Overall, across all transect stations, graminoid cover was not statistically significantly different between site and reference areas and did not correlate significantly with distance from the road (Tables 6-3 and 6-4). The need for further study of plant communities will be considered during development of the risk management plan.	Response is acceptable.
NPS-87	**Page 6-28. Zinc in Lichens.** Numerous studies have shown lichen declines related to zinc toxicity. Hasselbach et al. (2004) documented zinc levels of up to 2500 mg/kg. Other studies have shown lichen decline for zinc levels of only 200-600 mg/kg. It would be worthwhile sampling lichens at each of Hasselbach et al. (2004)'s sample points stratified by cover type to assess the effects of zinc on lichen communities here.	High	Please consider conducting the sampling work mentioned in this comment as part of future monitoring work at the site. See also recommendation for comment NPS-2.	Minimum zinc toxicity thresholds in lichen tissue reported in Folkeson and Andersson-Bringmark (1988) ranged from 480 μg/g for first signs of reductions in cover to 600 μg/g for first signs of mortality. These values correspond to the upper end of the toxicity range reported in this comment. Table CK2 (attached to this document and included in the revised risk assessment) provides a comparison of available lichen data with zinc effects thresholds from this study. Also, please see the responses to comments NPS-2 and NPS-16. The need for future study of plant communities (including lichens) will be evaluated during development of the risk management plan.	Response is acceptable.

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NPS-88	**Page 6-29**. **Lichen cover**. Lichen cover along the DMTS at 10, 100, and 1000 m was significantly lower than in Reference Area. Qualitative assessment showed that lichens were still lower than at the Reference Area at 2000 m. For most other taxa in the RA, the most significant effects apparently extended out to 100 m.	Medium	Please clearly indicate in the executive summary of the revised RA that effects on lichens have been observed up to 2 km from the haul road. In future monitoring work in the CAKR, please determine the full extent of impacts to lichens.	The Ecological Risk Assessment section of the executive summary now states: *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 up to 1,000 m from the road and 1,430 m from the port, may be a result of fugitive dust deposition; however, road effects or natural variability in plant communities may also be contributing factors for this observed difference. Further study would be required to verify the lichen results and to define the nature and extent of lichen effects related to fugitive dust deposition from the DMTS port and road and Red Dog Mine* The need for future study of plant communities (including lichens) will be evaluated during development of the risk management plan. In response to NPS-29 and this comment, the following sentence was added to the second bullet: *Differences between reference plant communities and plant communities beyond 1,000 to 2,000 m from the DMTS road, specifically the 2- to 4.5-fold decrease in lichen cover (Tables 6-10 and 6-11), may be a result of fugitive dust deposition. Further study would be required to verify the lichen results and to define the nature and extent of lichen effects related to fugitive dust deposition from the DMTS port and road and Red Dog Mine.*	Response is acceptable.
NPS-89	**Page 6-31. Excellent comment**. "Lichens may be eliminated entirely in areas of high dust and are the most affected growth form in the tundra…" Also, excellent observation that Sphagnum is harmed by Ca inputs—though this is not assessed quantitatively because mosses were not ID'ed to groups. It would be fruitful to compare Sphagnum levels in Reference Areas versus Site.	Medium	Please design future vegetation surveys so that relationships between Ca and Sphagnum at the site can be better understood. See NPS recommendations in comment NPS-21.	Comment noted. The need for future study of plant communities (including lichen and bryophyte species) will be evaluated during development of the risk management plan.	Response is acceptable.
NPS-90	**Page 6-32. Zinc**. Zn concentrations were reportedly high enough to cause mortality and/or reduction in cover up to 1000m from the DMTS road in feather mosses. That represents 64 km_2 in CAKR. It is also stated that zinc effects could extend up to 100m for lichens. This requires a great deal more study since lichens weren't identified to species and the sensitivities were based on only one study. Some species are much more tolerant than others to metal toxicity. The lichen literature is rich in studies on metals. Again, we also need to consider the acute effects that could occur during melt-off.	High	Please design future vegetation surveys so that relationships between metals and nonvascular species at the site can be better understood. See NPS recommendations in comment NPS-21.	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 due to 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 only used as supplemental lines of evidence to community evaluations in the risk characterization for terrestrial plants (Section 6.2.3). As noted in the comment, the moss and lichen phytotoxicity thresholds are based on a single study, which adds uncertainty to their application at the Red Dog site. The Folkeson and Andersson-Bringmark (1987) study was 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; relevant moss and lichen taxa (e.g., feather mosses and reindeer lichens); corresponding concentration and response gradients; comparable tissue data (metals concentrations in unwashed samples); and ecologically relevant endpoints, including survival and abundance (cover). No other study with these characteristics was identified in the literature review (although others would be considered, if available). Regarding potential for acute effects during	Response is acceptable.

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				snowmelt, some discussion is provided in the response to comment NPS-7. The need for future study of plant communities (including lichen and bryophyte species) will be evaluated during development of the risk management plan.	
NPS-91	**Page 6-33. Need for Long Term Monitoring Due to Large Affected Area.** Lichen cover is reported to be still 2 to 4.5 times lower than reference covers at 2000 m from the DMTS road. By extrapolation, this suggests that at least 128 km^2 of land is suffering reduced lichen cover. Given the scale of this observation, it appears we need considerably more study and mitigation. Long-term vegetation monitoring should be one component of these efforts, as it is easier to detect change over time than to control for differences at least in part caused by within and between plant community variability.	High	See recommendations for comments NPS-88, 89, and 90.	We agree with the commenter's point that monitoring over time will be an easier way to assess change than making comparisons between communities that have inherent variability. The need for future monitoring of plant communities (including lichen and bryophyte species) will be evaluated during development of the risk management plan.	Response is acceptable.
NPS-92	**Page 6-36. Cadmium and Lichens**. The one study cited shows that Cd is more toxic to lichens than Zn. If this bears out in the literature, then why is Zn being used for assessment of toxicity? In addition, since a zone up to 2000 m is strongly affected, additive effects from multiple stressors (Cd, Zn, S, SO_4) is probably the most likely scenario. As the RA has a regulatory approach, which appears to regulate each element separately, multiple causation is little considered.	Medium	Please add discussion of multiple causation to explain lichen impacts. Consider the chemicals named in this comment.	In the risk characterization for coastal plain and foothills mesic tussock tundra (Section 6.2.3.1), the text now states the following : *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.*	Response is acceptable.
NPS-93	**Page 6-36, Section 6.2.4 Soil Fauna.** The RA fails to evaluate effects to tundra soil fauna communities because ecological screening benchmarks are typically lower than for plants. We wonder to what extent the ORNL values reflect values in arctic Alaska. Some additional justification for omitting this receptor would be valuable.	Medium	Provide additional justification as requested in this comment.	Soil fauna were not evaluated directly in the Risk Assessment, but rather were evaluated assuming that if there were adverse effects due to presence of chemicals in tundra habitats, the effects would be apparent in plant communities. The basis for this approach is that ecological screening benchmarks for soil are typically much lower for plants than for soil fauna, and therefore, the results of the terrestrial plant community analysis would also be protective of the soil fauna community. This discussion was included in Section 6.2.4 (Risk Characterization for Tundra Soil Fauna), and is appended below: *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 due to the presence of chemicals in tundra habitats, these effects would be apparent in plant communities at concentrations where no effects would be seen on soil fauna. For this reason, it is assumed for purposes of the baseline risk assessment that results of the terrestrial plant community analysis will be protective of potential adverse effects to soil fauna. 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.* The extent to which ORNL values reflect screening benchmarks for tundra soil fauna communities is unknown, as the benchmarks are typically derived from species common to soils of more temperate regions of North America. However, as there are no screening benchmarks specific to tundra	Response is acceptable.

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				communities, the ORNL values represent the best information available for screening purposes. The discussion of ORNL values was included in Section 3.5.1 (Terrestrial Tundra Environment), and is appended below: *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.*	
NPS-94	**Page 6-51, 6-52. Lichens at Lagoon South.** NPS landcover maps show such tundra types as Sedge-Dryas Tundra, Crowberry Tundra, Partially Vegetated, Low Shrub Birch-Ericaceous Scrub around the Port Lagoon South. All of these cover types are favorable to high lichen diversity, far more so than the graminoid and tussock tundra communities sampled. Mosses and lichens were not collected at the lagoon. The conclusion that "coastal lagoon vegetation does not appear to be adversely affected" may be unwarranted given the high diversity of habitat types there and the presence of several habitat types known for high lichen diversity.	High	Please describe this shortcoming of the plant survey work in the revised ERA. See recommendations for comments NPS-88, 89, and 90.	To clarify the focus of the coastal lagoon plant community assessment to the reader, this additional text was added to the introduction to that section (Section 6.4.2): *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 overall terrestrial plant community assessment. Other vegetation types that may occur near the coast were not evaluated directly in this assessment.* The first sentence in the risk characterization (Section 6.4.2.4) now refers to the "*graminoid community surrounding coastal lagoons*" to distinguish these results from possible effects to other communities that were not directly assessed. Likewise, a clarifying statement was added to the concluding paragraph: *Note that 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 is uncertain.* This discussion has also been added to Section 6.6.4.3 (Uncertainty in Risk Characterization).	Response is acceptable.
NPS-95	**Sampling Transects on NPS Lands**. The only transect on NPS land is TT2. In future studies we would like to request that more attention be given to NPS lands affected by mining operations and transport.	Medium	Please include more transects on NPS lands in future vegetation monitoring studies at the site.	Please see response to comment NPS-24.	Response is acceptable.

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NPS-96	**Page 6-75, 6.5.4.1.1 Willow Ptarmigan.** If Pb continues to be distributed into the area along the DMTS over the next couple of decades and with the LOAELs close to 1.0 for ptarmigan, would not ptarmigan experience adverse affects such that precautionary tactics should be taken to minimize future release of fugitive dust? This is one example of why we disagree with Exponent that Teck-Cominco need not pursue any mitigation measures other than their "voluntary" clean-up efforts.	High	See recommendations for comments NPS-12 and 19.	Please see response to comment NPS-19. The ptarmigan findings as well as other risk assessment findings will certainly be considered carefully during the process of defining future actions to be taken, as part of the development of the risk management plan.	Response is acceptable.
NPS-97	**Page 6-76, 6.5.4.1.2 Tundra Vole.** If the terrestrial reference area were less enriched, would the LOAELs for Pb and other CoPCs change for tundra voles, showing a greater potential effect over background conditions?	Medium	Please provide an answer to this question in the revised ERA. See also recommendation for comment NPS-1.	The LOAELs would not change, as these are threshold effect levels derived from laboratory toxicity studies. However, if the CoPC concentrations were lower in the reference area, the reference area hazard quotients would decrease, since exposure would be lower relative to the LOAEL. In this case, there would be an increase in the relative magnitude of hazard quotients for tundra voles at the site relative to the reference area (assuming no concurrent change in CoPC concentrations at the site),	Response is acceptable.
NPS-98	**Page 6-76. NPS Risk Tolerance**. The RA states that risk to tundra voles doesn't translate into an "unacceptable ecological risk to the site's vole population as a whole…" From NPS's perspective, the population as a whole does not need to be threatened before we become concerned about the level of impacts. The AIDEA easement through CAKR ranges from about 140 m to 3 km from the DMTS haul road, therefore the effects described for vegetation are already having impacts on CAKR lands both inside and outside the easement boundaries.	High	See recommendation for comment NPS-12.	Please see the response to comment NPS-12 regarding the potential for population-level effects to wildlife.	Response is acceptable.
NPS-99	**Page 6-77, 6.5.4.1.3 Caribou.** Though we agree the entire population of the WACH is unlikely to be affected by CoPCs from the Red Dog mine because they migrate so far and so fast, we are more concerned with sub-populations that remain near the DMTS facilities. We understand that as many as 200 caribou stay near the mine during some winters, consuming fugitive dust-contaminated lichens all winter. Again, the RA emphasis on the huge range and population of the WACH minimizes appropriate concern about smaller populations of caribou or other wildlife that use habitat with heavy metals enrichment along the DMTS. Though this approach put things in a regional context for caribou, the data about heavy metals enrichment along the DMTS do not support Exponent's suggestion that further actions are not required to change the trend of increasing metals enrichment along the DMTS.	High	See recommendations for comments NPS-12, 19, and 20.	The risk assessment conclusions in this section clearly distinguish between the potential magnitude of effects to migratory caribou and seasonally resident caribou, and notes that adverse effects may potentially occur to seasonal residents from exposure to aluminum. Regarding further actions to change the trend of metals enrichment, please refer to the response to comment NPS-19.	Response is acceptable.
NPS-100	**Page 6-82, 6.6.2 Uncertainties Related to Terrestrial Assessment.** As noted above, an additional uncertainty for plant assessment would be the number of plots sampled to achieve statistical validity in describing vegetation variation.	Medium	Please expand the uncertainty discussion as requested in this comment.	Uncertainties related to sample size are now discussed in Section 6.6.4.1.4. The text related to the number of microplots sampled is included below: *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 ME1. Each graph summarizes the results for one plant*	Response is acceptable.

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				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. *The species-area curves for the coastal plain community suggest that ten microplots were sufficient to capture most species (Figure ME1). 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 6-14), 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 6-29). 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 6-29). 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 6-14).* *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*	

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				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.	
NPS-101	**Page 6-83, Section 6.6.2.1. Monitoring.** Monitoring for vegetation on all of the major landcover types—with special emphasis on those high in nonvascular plant cover and diversity—should be a high priority for upcoming work.	Medium	Please consider this suggestion when designing the risk management plan at the site.	The need for future study of plant communities (including nonvascular species) will be an important topic of consideration during development of the risk management plan.	Response is acceptable.
NPS-102	**Page 6-84. Selection of Reference Areas**. Two major questions remain about the choice of Reference Areas: 1) Why were there only two Reference Areas? 2) Why weren't they located in a zone clearly outside the influence of fugitive dust (Hasselbach et al. 2004) or mineralization (Fig 1-4, Table 3-4).	High	See recommendation for comment NPS-1.	Please see response to comment NPS-1.	Response is acceptable.
NPS-103	**Page 6-85. Vegetation Cover Estimation**. Ecologists often estimate cover for each species separately, even if in the understory, such that the total cover on a plot can be >100%. This more closely approximates the biomass on the plot than 100%-based dominant canopy cover estimates.	Medium	In the revised ERA, please discuss the impact of this alternative approach for estimating biomass on the results and conclusions of the vegetation survey.	Comment noted. Both are established methods for estimating percent cover (Barbour et al. 1980). As explained in the methods (Section 6.2.1.1), "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," rather than a surrogate for biomass. Uncertainties surrounding the interpretation of cover estimates are discussed in Section 6.6.4.1.3 (Field Sampling Methods).	Response is acceptable.
NPS-104	**Page 6-86, second paragraph. Mosses and Lichens Evaluated at Group Levels**. In the RA, mosses and lichens were treated as one group each rather than being treated at the species level. As noted, tolerances to metals toxicity varies widely among nonvascular taxa. Follow up study should focus on species, and should use nonmineralized, clean reference areas.	High	Please follow these suggestions when designing future vegetation monitoring studies at the site.	Comment noted. The need for future study of plant communities (including nonvascular species) will be evaluated during development of the risk management plan. Consideration will be given to possible selection of alternate reference areas.	Response is acceptable.
NPS-105	**Page 6-86, Section 6.6.2.2. Uncertainty in Comparisons to Phytotoxicity Thresholds.** A recent literature search on the lichen literature search engine (http://www.nhm.uio.no/botanisk/lav/RLL/RLL.HTM) generated a list of 44 publications just for zinc and 269 publications for heavy metals. The RA bases its evaluation of phytotoxicity on 2 studies. One of these is a field study of a coniferous woodland community near a brass foundry, which is not strikingly similar to tundra communities exposed to Zn, Cd and S. Since mosses	High	See recommendations for comments NPS 2, 6, and 19.	Comment noted. Please see the responses to comments NPS-2, NPS-6, NPS-19, and NPS-90.	Response is acceptable.

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	and lichens were not identified to species an evaluation of species sensitivity and adverse impacts could not be assessed with this project. Because there is evidence that mosses and lichens are adversely affected at considerable distances from the DMTS corridor, we believe it is incorrect to assume no further actions should be taken by the industry to reduce fugitive dust emissions along the DMTS. We agree with the author's statement on page 6-87, section 6.6.2.3 that further study is needed to elucidate the role of CoPCs from the DMTS relative to other road effects (e.g., hydrological effects, road dust, dust palliatives).				
NPS-106	**Page 6-87, First Paragraph. Retention of Dust on Samples**. It is stated that plant samples weren't washed before analysis. We are a bit unclear on exact sample handling however. Were the specimens analyzed together with the dust remaining in the bottom of the sample bags— so that the full amount of dust originally on the leaves was included? Were the plants shaken in the field to remove dust prior to bagging? Some discussion would be helpful.	Low	Please provide the requested information in the revised ERA.	Section 6.2.2 and in Section 6.2, where terrestrial methodology is discussed, has been revised to clarify that plant tissue samples were neither shaken nor washed before chemical analysis (this is the first place in the text where comparisons with phytotoxicity thresholds are discussed). The analytical laboratory tested the entire contents of the sample containers, including plant tissues and any loose dust.	Response is acceptable.
NPS-107	**Page 6-89, 6.6.3.1.1 Body Masses and Intake Rate Parameters.** Because the models use average size individuals in a receptor wildlife population, they tend to underestimate exposures to smaller members of a population and overestimate exposure to larger members. If effect levels are reached, we think smaller members of a population are likely to be selected against, thereby potentially affecting the genetic make-up of a population.	Medium	In the revised ERA, please indicate how large an effect body mass and intake rate have on the wildlife risk estimates. Use example calculations as appropriate.	The uncertainty analysis in the second paragraph of section 6.6.5.1.1 has been expanded to include a discussion of what effect using average body mass has on wildlife risk estimates, and that text is included below: *For many receptors, average male body mass may be higher than that of females, but since food ingestion rates scale with body weight and since 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 6-26). Since 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.*	Response is acceptable.

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NPS-108	**Page 6-92, 6.6.3.3 Representativeness of Sampling Locations.** This section states that tissue data from wildlife receptors were adequate to detect spatial patterns relative to the DMTS, including a return to background levels of risk. We maintain all reference sampling locations are enriched with heavy metals above background from the DMTS. This study cannot properly determine return to background levels because no samples were obtained from areas unaffected by DMTS fugitive dust.	High	See recommendation for comment NPS-1.	Please see response to comment NPS-1.	Response is acceptable.
NPS-109	**Page 6-96, 6.6.3.5, Uncertainty in TRV Extrapolation.** We are concerned that the hazard quotients for birds exceed 1.0 for mercury and zinc at all stations, *including the reference area*. This is understandable for mercury because this heavy metal is very volatile and is transported to and concentrated in arctic regions from global emissions. Zn is more likely derived from fugitive dust emissions from the DMTS *including to reference sites* and is less likely derived from deeply buried and bio*unavailable* bedrock.	High	Based on information provided by the NPS, please determine the potential bias in the background risk estimates for mercury and zinc based on the NPS claim that the reference areas do not represent true background. Summarize the findings in the revised ERA. See also recommendations for comment NPS-1.	Please see response to comment NPS-1. The additional uncertainty discussion that has been added to Section 6.6 is included in that response.	Response is acceptable.
NPS-110	**Page 6-98, Last Paragraph. Ecological Significance**. It is argued in the RA that there is a marked decline in lichens that's related to distance from the DMTS road and that the effect continues beyond 2000 m. Exponent writes in assessing overall significance that "the adverse effects are most pronounced in the first 100 m and are not expected to occur at any substantial distance from the road, port or mine…" NPS considers lichens to be highly significant members of their ecosystem in terms of both forage and diversity. This statement understates the importance of nonvascular plants in the arctic ecosystem.	High	See recommendation for comment NPS-6.	In general, ecological effects were most pronounced near dust sources at the port, road, and mine, and for most receptors, there was a low potential for effects at 1,000-m stations. However, lichen effects were detected at stations beyond 100 m. Therefore, the following text was added to the last paragraph of Section 6.7.1 (Terrestrial Habitats): In summary, the potential for adverse effects to wildlife is most pronounced in the first 100 m adjacent to the road or facilities (Table JS5b) and effects in general are not expected to occur at any substantial distance from the road, port facilities, or mine ambient air/solid waste boundary. However, lichen cover values at 1,000-m and 2,000-m stations were significantly lower than reference cover values, suggesting that lichen effects may still occur at these distances from the DMTS road corridor. Furthermore, 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 verify the lichen effects observed at distances greater than 100 m from dust sources, and possibly beyond 2,000 m, to define the nature and extent of these effects on lichens, and to distinguish the relative contributions of causative agents, such as metals and road dust or other factors, on lichen toxicity. Overall, results of the ERA show that adverse effects to the terrestrial habitats and receptors are largely restricted to localized areas adjacent to the DMTS road, the port facility, and the mine ambient air/solid waste boundary, as summarized in Table JS7.	Response is acceptable.
NPS-111	**Page 6-98, Last Paragraph. Metals vs. Normal Road Dust.** Exponent writes that "the contribution of metals in producing some of these effects, particularly on plant communities near the DMTS road, is unclear." Auerbach (1997) concludes that distance to the Dalton Highway in arctic Alaska is correlated at only r^2=0.28 and r^2=0.08 with lichen biomass at two different study areas with vegetation similar to that near the DMTS road. Table 6-4 shows a correlation of 0.77 between distance to DMTS road and lichen cover.	High	In the revised ERA, alternative explanations for lichen decline should be rigorously evaluated. The discussion should address the specific points made in this comment.	The commenter refers to regression results for lichen biomass presented in Table 1 of Auerbach et al. (1997). In Table 2 of the same article, the authors also present results of regressing lichen species' covers on log of distance from the Dalton Highway; significant r^2 values ranged from 0.13 to 0.42. In both cases, r^2 values reported in Auerbach et al. (1997) for the Dalton Highway 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)	Response is acceptable.

No.	Comment	Priority	Recommendation	Response	DEC Remarks
				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. In response to this comment, the following discussion was added to Section 6.6.4.3. (Uncertainty in Risk Characterization): *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.* In addition, the following sentence was added to the last paragraph of Section 6.2.3.1.1. (Conclusions): *Further study would be required to verify the lichen results and to define the nature and extent of lichen effects related to fugitive dust deposition from the DMTS port, road, and Red Dog Mine.* Please also see comment response to NPS-110.	
NPS-112	**Page 7-1, Section 7.2 Ecological Risk Based Action Levels**. It is again stated that effects to terrestrial vegetation may simply be a function of normal road dust. In light, both of the preceding comment on road dust and the known toxicity of Zn, Cd and S to lichens, this statement appears to demonstrate a strong bias. Clearly this entire topic warrants considerably more study in addition to a suite of effective contaminant control measures. The statement suggesting no action levels are required because the role of metals cannot be quantified is inadequate because it cannot be demonstrated CoPC metals were not responsible for ecological changes and stress. These questions cannot be answered with the level of study conducted to date and we disagree with the summary judgment. It would be more accurate to	High	See recommendations for comments NPS-6, 19, and 111. Adverse impacts, whether due to normal road dust or metals, should still be identified as adverse impacts in the revised ERA.	Ecological risk assessment conclusions have been modified in response to these comments. For example, in Section 7.2 (Ecological Risk Based Action Levels), the following sentences have been added to the end of the second paragraph: *Collectively, results for wildlife receptors indicate that population-level impacts are unlikely to occur; indicating that calculation of risk based levels may be unnecessary. However, the potential use of ecologically based action levels will be evaluated further in development of the risk management plan, as described in Section 7.3 (Risk Management Plan).* Also in response to this comment, the highlighted changes have been made to Section 7.3 (Risk Management Plan):	Response is acceptable.

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	state more study is required before a judgment can be made whether to assign action levels to further control fugitive dust along the DMTS to prevent or reduce adverse effects to tundra vegetation. We also believe it's also overly simplistic to state that since no action is warranted because no one CoPC is responsible. First, with regard to lichens, we have no proof that this is true, and we have strong suspicion about Zn, Cd and S. Second, the concept of additive stressors needs to enter this equation somehow. If the multiple effects of two CoPC's cause injury or mortality to an organism—but neither can produce as strong of an effect alone—it makes little sense to claim that no action is required because these elements are below certain effects thresholds. Lastly, the RA bases the inappropriateness of action levels because of a limited spatial scale. As noted, the minimum size of the affected area of nonvascular vegetation is 128 km_2. At what point does a "limited scale" become a sizeable area? In addition, does an entire population need to be threatened before action is warranted, or would action be warranted if demonstrable effects to small mammals on 6 km_2 are shown?			*A risk management plan will be developed to address the issues identified by the risk assessment. The plan will include evaluation of risk management options within the general categories of institutional and engineering controls to control current sources of fugitive dust, monitoring, and remediation/restoration. The plan will identify the most appropriate combination of actions for management of risk, over the life of the mine.* *As described previously, human health based action levels were not calculated at this time because human health risks are not significantly elevated. However, some ecological risks were identified, as described in Section 7.2. Human health or ecologically based action levels could be used as one component of a risk management strategy, e.g., as a tool for risk management associated with monitoring and/or with Teck Cominco's voluntary cleanup program. The potential need for and use of action levels will be further evaluated in the process of developing the risk management plan. If numerical action levels are determined to be needed, they will be calculated and included in the plan.* -- Please also see the response to comment NPS-19 regarding calculation of action levels and additional actions that may be required to address risks identified by the risk assessment. -- The revised ecological risk assessment conclusions are included in the risk assessment, and the revised text from Section 8 (Conclusions) is provided below: First, the introductory paragraph was revised as follows: *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. The following subsections summarize the findings of the human health and ecological risk assessments.* In Section 8.2.1, the second, third, and fifth bullets have been revised: • *Bullet 2: Differences between reference plant communities and plant communities beyond 1,000 to 2,000 m from the DMTS road, specifically the 2- to 4.5-fold decrease in lichen cover (Table 6-10 and 6-11), may be a result of fugitive dust deposition. Further study would be required to verify the lichen results and to define the nature and extent of lichen effects related to fugitive dust deposition from the DMTS port and road and Red Dog Mine.* • *Bullet 3: 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.* • *Bullet 5: Adverse effects to herbivorous birds (e.g., ptarmigan) are possible in populations near the port and mine, particularly the most highly exposed individuals. These effects, if occurring, could result in*	

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				population-level effects in areas near the port or mine. However, the likelihood of adverse effects to herbivorous birds foraging along the length of the road is low, as 95 percent UCL on the mean exposures did not exceed NOAEL and/or LOAEL TRVs in these areas. The second bullet in Section 8.2.4 (Coastal Lagoons) has been revised: • *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. Extrapolation of these results to other coastal lagoon communities is uncertain.*	
NPS-113	**Page 7-2, 7.2 Ecological Risk Based Action Levels.** Similarly, the conclusion that no action levels should be required to reduce exposures and potential impacts to small mammals and birds along the DMTS is in error because the reference sites are within the zone of fugitive dust deposition and calculations of hazard quotients at study sites relative to reference sites are not as far apart as they should be. In our opinion the conclusions are invalidated by used of impacted reference sites that are assumed to be reasonable indicators of uncontaminated, natural background levels.	High	See recommendations for comments NPS-1, 12, and 19.	Please see responses to comments NPS-1, NPS12, and NPS-19. The text in Sections 7.2 and 7.3 has been revised in response to this and other comments, and the revised text is outlined above in response to Comment NPS-112.	Response is acceptable.
NPS-114	**Page 8-2 and 8-3. Conclusion—Plant Community Structure**. We are unclear as to why it is concluded that there are changes in vegetation community structure within 100 m of the road and port when elsewhere in the RA it clearly states that effects to lichens extend beyond 2000 m of the DMTS road.	High	See recommendation for comment NPS-6.	The plant community surveys support the conclusion that plant community structure is altered near dust sources. The second bullet in Section 8.2.1 (Terrestrial Habitats) addresses vegetation effects beyond 100 m from dust sources. This second bullet has been revised to state: *Differences between reference plant communities and plant communities beyond 1,000 to 2,000 m from the DMTS road, specifically the 2- to 4.5-fold decrease in lichen cover (Table 6-10 and 6-11), may be a result of fugitive dust deposition. Further study would be required to verify the lichen results and to define the nature and extent of lichen effects related to fugitive dust deposition from the DMTS port and road and Red Dog Mine.*	Response is acceptable.
NPS-115	**Page 8-3, 8.2.1 Ecological Risk Assessment for Terrestrial Habitats, Bullet 3.** The authors should research CSB1 construction history before suggesting or dismissing vegetative impacts downwind from the structure as being caused by the construction and its subsequent effects (to hydrology or other factors.) This situation begs more study; it does not reject possible effects from CoPCs.	High	Please remove the conjecture in this section of the report.	The third bullet was revised to state: *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. 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.*	Response is acceptable.
NPS-116	**Page 8-3, 8.2.1 Ecological Risk Assessment for Terrestrial Habitats, Bullet 4.** The last statement in this section indicates population level effects to small mammals are unlikely because of the limited spatial scale of effects and the uncertainties associated with TRVs, but the discussion on TRVs indicates these values could be low or high and are based on different species in different habitats.	High	See recommendation for comment NPS-12.	Please see the response to comment NPS-12 for discussion of uncertainty and conclusions regarding the potential for population-level effects. No changes were made to the fourth bullet in Section 8.2.1. Please refer to the uncertainty discussion on TRVs in Section 6.6.5.4 (Toxicity Reference Values), which includes information outlining why the aluminum and barium TRVs are considered conservative and likely overestimate risks to small	Response is acceptable.

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				mammals. Revisions were made to the Interpretation of Ecological Significance, Section 6.7.1, as shown below: *Herbivorous small mammals (i.e., tundra vole and tundra shrew) inhabiting tundra within 10-100 m of the DMTS road near the port facilities or near the mine's ambient air/solid waste boundary (i.e., along transects TT6 and TT7) showed incremental risk from exposure to barium, and aluminum. 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. Therefore, if adverse effects occur to 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 or barium.* *Regardless, possible effects on individuals in these areas, such as reduced growth (the endpoint for the aluminum TRVs) or increased mortality (the endpoint for the barium LOAEL TRV), are unlikely to translate into regional population-level effects given the limited area where adverse effects could occur, uncertainties related to the derivation of aluminum and barium TRVs, and extrapolation of individual-level responses to population endpoints, as discussed above in Section 6.6. In addition, aluminum and barium TRVs were derived from studies using much more soluble and bioavailable forms of barium and aluminum than those found at the site. Also, the barium endpoints for mammals based on rat studies using these more bioavailable forms (i.e., hypertension for the NOAEL, increased kidney masses and reduced ovarian masses for the LOAEL) are not conclusive as to their potential for effects on the populations. For aluminum, no effects have been found in avian studies, and in mammalian studies, the only effects endpoint was a reduction in weight gain of offspring in the second and third litters of second- and third-generation mice.* *Aluminum and barium are therefore not expected to be the risk drivers, as a result of the low solubility and low bioavailability of the forms present on the site. This was also illustrated in recent bioaccessibility testing work (Shock et al. 2007). The results of that research 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. In the risk assessment described throughout this document, the bioavailability of metals in soils was assumed to be 100 percent.*	
NPS-117	**Page 8-4**. **Conclusion—Plant Communities at Port Lagoon**. As noted earlier, an inadequate number of plant community types were sampled in this effort. Specifically omitted were those types hosting high diversity of lichen taxa. This conclusion would probably not withstand the scrutiny of detailed study.	High	Please describe this shortcoming of the plant survey work in the revised ERA. Discuss the effect it has on the conclusions drawn for vegetation impacts at the lagoons. See recommendations for comments NPS-88, 89, and 90.	Please see the response to comment NPS-94. Clarifying language was also added to Section 6.7.3 (Interpretation of Ecological Significance). Section 6.7.3 is included below, and the revised text is highlighted in blue: *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 may not be directly applicable to other coastal plant communities with different*	Response is acceptable.

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				compositions. Food web models 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 due to 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. Also based on the modified text in 6.7.3, and in response to this comment, the second bullet of Section 8.2.4 (Coastal Lagoon Conclusions) was modified. Revisions are highlighted below: • *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. Extrapolation of these results to other coastal lagoon communities is uncertain.*	
NPS-118	**Effects of Road Dust vs. Concentrate**. The RA suggests that the effects of road bed material dust on vegetation cannot be distinguished from the effects of ore concentrate dust. We acknowledge that these substances are mixed by wind and traffic and that the relative importance of inert physical properties vs. toxicity is not readily discernable by monitoring. However, their combined impacts can be measured and remain important. We also believe that experimentation could distinguish between road and ore concentrate effects.	Medium	See recommendations for comments NPS-6, 19, and 111. Adverse impacts, whether due to normal road dust or metals, should still be identified as adverse impacts in the revised ERA.	Please see the responses to comments NPS-6, NPS-19, and NPS-111. Vegetation effects, regardless of the ultimate cause, are identified in the Terrestrial Assessment (Section 6.2) and acknowledged in the Interpretation of Ecological Significance (Section 6.7).	Response is acceptable.

Notes: Please note that RA text quoted herein may differ from that in other comment response documents, and in comparison with the final RA document, as a result of successive revisions made during the comment resolution process.

NPS comments were prepared by the NPS-Western Arctic National Parklands in collaboration with NPS Alaska Regional Office. Collaborating Staff: Peter Neitlich, Bud Rice, Linda Hasselbach, and Robert Winfree.

See original NPS comment letter for complete citations of cited literature.

ADPH - Alaska Department of Public Health
ALM - adult lead model
CAKR - Cape Krusenstern National Monument
COPC - chemical of potential concern
CSB - concentrate storage building
DEC - Department of Environmental Conservation (Alaska)
DMTS - DeLong Mountain Regional Transportation System
EPC - exposure point concentration
E&E - Ecology and Environment, Inc.
ERA - ecological risk assessment
IEUBK - integrated exposure uptake/biokinetic model
LOAEL - lowest-observed-adverse-effect level
NA - not applicable
NANA - Northwest Arctic Native Association
NMS or NMDS - nonmetric multi-dimensional scaling
NPS - National Park Service
PCA - principal component analysis
RA - risk assessment
TC - Teck Cominco
TRV - toxicity reference value
USGS - United States Geological Survey.

Table 2-5. Relative importance of potential human exposure pathways[a]

Metal	Human Exposure Pathways: Inhalation	Dermal	Ingestion	Cumulative PRG
Aluminum	2,882,040		78,214	76,142
Antimony			31	31
Arsenic (cancer)	588	4.5	0.43	0.39
Arsenic (noncancer)		279	23	21.6
Barium	294,086		5,475	5,375
Cadmium (cancer)	1,405			1,404
Cadmium (noncancer)		698	39	37
Chromium VI (cancer)	30			30
Chromium VI (noncancer)	4,529		235	223
Cobalt (cancer)	903			903
Cobalt (noncancer)	11,734		1,564	1,380
Copper			3,129	3,129
Fluoride		16,760	4,693	3,666
Iron			23,464	23,463
Lead[b]				
Manganese	28,820		1,877	1,762
Mercury			23	23
Molybdenum			391	391
Nickel			1,564	1,564
Selenium			391	391
Silver			391	391
Strontium			46,929	46,924
Thallium			5.2	5.2
Tin			46,929	46,924
Vanadium			78	78
Zinc			23,464	23,463

Note: Units are in mg/kg.

EPA - U.S. Environmental Protection Agency
PRG - preliminary remediation goal

[a] The screening values listed above are U.S. EPA (2006c) Region 9 PRGs for residential soil. This table is not meant to provide screening concentrations for the DMTS risk assessment. Rather, the PRGs listed above are provided to illustrate the relative contribution of inhalation, dermal contact, and ingestion exposure. The PRGs were derived assuming a risk level of 1×10^{-6} for cancer and a hazard quotient of 1.0 for noncancer endpoints. Higher PRGs indicate relatively lower contribution to risk, and vice versa. These PRGs suggest that dermal contact is at least an order of magnitude lower risk than ingestion, and that inhalation is several orders of magnitude lower risk than ingestion.

[b] Lead risks are evaluated using separate models that do not predict a hazard quotient, thus they are not directly comparable to risks from other metals.

Table ME1. Correlation of NMDS axes with environmental variables

	All Vegetation Communities				Coastal and Tundra Only			
	Axis 1		Axis 2		Axis 1		Axis 2	
	Correlation	*p*-Value	Correlation	*p*-Value	Correlation	*p*-Value	Correlation	*p*-Value
Distance	**-0.50**	**0.0812**	**-0.51**	**0.0794**	**-0.61**	**0.0665**	**-0.84**	**0.0045**
Antimony	0.19	0.3973	0.35	0.1146	0.12	0.7028	**0.81**	**0.0013**
Arsenic	-0.19	0.3957	0.04	0.8767	0.37	0.2059	**0.69**	**0.0103**
Barium	**0.54**	**0.0125**	0.05	0.8192	**0.54**	**0.0611**	**0.63**	**0.0237**
Cadmium	0.14	0.5484	0.30	0.1811	0.08	0.7855	**0.75**	**0.0046**
Cobalt	0.07	0.7561	0.13	0.5637	0.45	0.1272	0.36	0.2277
Copper	0.14	0.5447	0.04	0.8745	0.41	0.1604	**0.64**	**0.0202**
Lead	0.12	0.5870	0.35	0.1174	0.30	0.3200	**0.77**	**0.0028**
Manganese	**0.66**	**0.0015**	0.03	0.8923	**0.53**	**0.0657**	0.47	0.1035
Mercury	0.04	0.8745	**0.37**	**0.0962**	-0.25	0.4150	**0.61**	**0.0294**
Molybdenum	-0.18	0.4352	**-0.45**	**0.0441**	0.28	0.3534	-0.19	0.5292
Selenium	-0.05	0.8456	0.24	0.2941	0.15	0.6298	**0.62**	**0.0261**
Silver	0.23	0.3119	0.25	0.2668	0.22	0.4647	**0.82**	**0.0011**
Thallium	**0.40**	**0.0730**	-0.02	0.9348	0.31	0.2929	**0.73**	**0.0061**
Vanadium	-0.10	0.6737	-0.13	0.5695	**0.59**	**0.0350**	**0.50**	**0.0831**
Zinc	0.11	0.6430	**0.43**	**0.0535**	0.14	0.6494	**0.88**	**0.0000**
pH	0.31	0.1697	0.27	0.2285	0.40	0.1695	**0.67**	**0.0139**
Total Solids	**0.39**	**0.0829**	0.08	0.7347	0.37	0.2130	**0.76**	**0.0034**

Note: Spearman rank non-parametric method was used.

Bold entries indicate significant correlation (p<0.10).

Table 6-9. Summary of correlations between rotated PCA factors and distance or soil characteristics

	All Vegetation Communities				Coastal[a] and Tundra[b] Communties only			
	Factor 1		Factor 2		Factor 1		Factor 2	
	Correlation	*P*-value	Correlation	*P*-value	Correlation	*P*-value	Correlation	*P*-value
Distance	-0.178	0.5323	**0.521**	**0.0724**	-0.303	0.3546	**0.883**	**0.0085**
Soil Parameters								
Antimony	0.156	0.4876	-0.309	0.1660	-0.412	0.1507	**-0.484**	**0.0921**
Arsenic	-0.309	0.1666	**-0.758**	**0.0007**	-0.041	0.8787	**-0.760**	**0.0082**
Barium	**0.384**	**0.0865**	**-0.503**	**0.0244**	-0.036	0.8939	**-0.721**	**0.0122**
Cadmium	0.101	0.6526	**-0.425**	**0.0572**	-0.346	0.2268	**-0.538**	**0.0608**
Cobalt	-0.075	0.7340	**-0.451**	**0.0436**	0.203	0.4872	**-0.615**	**0.0323**
Copper	0.083	0.7123	**-0.800**	**0.0003**	-0.016	0.9469	**-0.747**	**0.0094**
Lead	0.042	0.8548	**-0.539**	**0.0158**	-0.088	0.7535	**-0.709**	**0.0137**
Manganese	**0.522**	**0.0197**	-0.283	0.2044	0.187	0.5237	**-0.632**	**0.0279**
Mercury	0.140	0.5320	-0.086	0.6969	-0.258	0.3659	-0.060	0.8267
Molybdenum	-0.227	0.3091	**-0.445**	**0.0462**	-0.245	0.3904	0.135	0.6469
Selenium	-0.186	0.4047	**-0.476**	**0.0330**	-0.344	0.2292	-0.410	0.1523
Silver	0.155	0.4912	**-0.574**	**0.0102**	-0.264	0.3553	**-0.635**	**0.0270**
Thallium	0.296	0.1864	**-0.661**	**0.0031**	-0.137	0.6274	**-0.725**	**0.0117**
Vanadium	-0.303	0.1750	**-0.734**	**0.0010**	-0.099	0.7247	**-0.670**	**0.0197**
Zinc	0.060	0.7916	**-0.430**	**0.0542**	-0.126	0.6547	**-0.643**	**0.0253**
pH	0.003	0.9930	**-0.472**	**0.0345**	-0.379	0.1856	**-0.522**	**0.0693**
Total solids	0.223	0.3205	-0.309	0.1667	0.066	0.8267	**-0.747**	**0.0094**

Note: **Bold** values indicate significant correlation at $p < 0.10$.
Spearman rank non-parametric correlation was used to estimate correlations.
PCA - principal component analysis

[a] Coastal plain mesic tussock tundra community.

[b] Foothills mesic tussock tundra community.

Table CK1. Comparison of tissue threshold concentrations in moss samples (*Hylocomium splendens*)

	Station	Zone	Sample ID	Event	Copper	Tissue Threshold Concentrations[a]	Zinc	Tissue Threshold Concentrations[a]
					mg/kg	A = 25 - 60	μg/g	A = 150 - 290
					dry	B = 35 - 90	dry	B = 190 - 350
						C = 70 - 110		C = 300 - 400
Site								
	001P-M01	ECO-R	001P-M-01	2001			1530	C
	002P-M01	ECO-R	002P-M-01	2001			1970	C
	003P-M01	ECO-R	003P-M-01	2001			2060	C
	004P-M01	ECO-R	004P-M-01	2001			1420	C
	005P-M01	ECO-R	005P-M-01	2001			2090	C
	006P-M01	ECO-R	006P-M-01	2001			1970	C
	007P-M01	ECO-R	007P-M-01	2001			1280	C
	008P-M01	ECO-R	008P-M-01	2001			1330	C
	009D-M01	ECO-R	009D-M-01	2001			3440	C
	009P-M01	ECO-R	009P-M-01	2001			3210	C
	010P-M01	ECO-R	010P-M-01	2001			2490	C
	011P-M01	ECO-R	011P-M-01	2001			1110	C
	013P-M01	ECO-R	013P-M-01	2001			1450	C
	015P-M01	ECO-R	015P-M-01	2001			424	C
	016P-M01	ECO-R	016P-M-01	2001			1160	C
	017P-M01	ECO-R	017P-M-01	2001			191	B
	018D-M01	ECO-R	018D-M-01	2001			261	B
	018P-M01	ECO-R	018P-M-01	2001			264	B
	019P-M01	ECO-R	019P-M-01	2001			518	C
	020P-M01	ECO-R	020P-M-01	2001			901	C
	021P-M01	ECO-R	021P-M-01	2001			1250	C
	022P-M01	ECO-R	022P-M-01	2001			602	C
	023P-M01	ECO-R	023P-M-01	2001			981	C
	024P-M01	ECO-R	024P-M-01	2001			1140	C
	025P-M01	ECO-R	025P-M-01	2001			862	C
	026D-M01	ECO-R	026D-M-01	2001			420	C
	026P-M01	ECO-R	026P-M-01	2001			290	B
	028P-M01	ECO-R	028P-M-01	2001			922	C
	029P-M01	ECO-R	029P-M-01	2001			119	
	030P-M01	ECO-R	030P-M-01	2001			209	B
	030R-M01	ECO-R	030R-M-01	2001			124	
	031P-M01	ECO-R	031P-M-01	2001			301	C
	031R-M01	ECO-R	031R-M-01	2001			348	C
	032P-M01	ECO-R	032P-M-01	2001			207	B
	032R-M01	ECO-R	032R-M-01	2001			169	A
	033P-M01	ECO-R	033P-M-01	2001			117	
	034D-M01	ECO-R	034D-M-01	2001			93.6	
	034P-M01	ECO-R	034P-M-01	2001			109	
	034R-M01	ECO-R	034R-M-01	2001			97.3	
	035P-M01	ECO-R	035P-M-01	2001			92.5	
	036P-M01	ECO-R	036P-M-01	2001			559	C
	036R-M01	ECO-R	036R-M-01	2001			436	C
	037P-M01	ECO-R	037P-M-01	2001			179	A
	038P-M01	ECO-R	038P-M-01	2001			116	
	038R-M01	ECO-R	038R-M-01	2001			153	A
	039P-M01	ECO-R	039P-M-01	2001			187	A
	040P-M01	ECO-R	040P-M-01	2001			72.3	
	040R-M01	ECO-R	040R-M-01	2001			71.9	

Table CK1. (cont.)

Station	Zone	Sample ID	Event	Copper	Tissue Threshold Concentrations[a]	Zinc	Tissue Threshold Concentrations[a]
				mg/kg dry	A = 25 - 60 B = 35 - 90 C = 70 - 110	μg/g dry	A = 150 - 290 B = 190 - 350 C = 300 - 400
041P-M01	ECO-R	041P-M-01	2001			309	C
042D-M01	ECO-R	042D-M-01	2001			84.2	
042P-M01	ECO-R	042P-M-01	2001			83	
042R-M01	ECO-R	042R-M-01	2001			82.9	
044P-M01	ECO-R	044P-M-01	2001			230	B
044R-M01	ECO-R	044R-M-01	2001			184	A
045P-M01	ECO-R	045P-M-01	2001			74.4	
046P-M01	ECO-R	046P-M-01	2001			223	B
048P-M01	ECO-R	048P-M-01	2001			129	
048R-M01	ECO-R	048R-M-01	2001			148	
050P-M01	ECO-P	050P-M-01	2001			377	C
051A-M01	ECO-P	051A-M-01	2001			358	C
052P-M01	ECO-P	052P-M-01	2001			637	C
053D-M01	ECO-P	053D-M-01	2001			197	B
053P-M01	ECO-P	053P-M-01	2001			193	B
059D-M01	ECO-P	059D-M-01	2001			300	B
059P-M01	ECO-P	059P-M-01	2001			384	C
060P-M01	ECO-P	060P-M-01	2001			340	C
102P-M01	ECO-R	102P-M-01	2001			141	
103P-M01	ECO-R	103P-M-01	2001			85.6	
116P-M01	ECO-R	116P-M-01	2001			87.8	
117P-M01	ECO-R	117P-M-01	2001			101	
117R-M01	ECO-R	117R-M-01	2001			119	
161P-M01	ECO-P	161P-M-01	2001			128	
161R-M01	ECO-P	161R-M-01	2001			156	A
201P-M01	ECO-R	201P-M-01	2001			132	
HR01-01A	ECO-P	HR-01-01-M	2001			4180	C
HR01-02M	ECO-P	HR-01-02-M	2001			2040	C
HR01-03M	ECO-P	HR-01-03-M	2001			273	B
HR02-01M	ECO-P	HR-02-01-M	2001			3140	C
HR02-02M	ECO-P	HR-02-02-M	2001			949	C
HR02-03M	ECO-P	HR-02-03-M	2001			59.2	
HR03-01M	ECO-R	HR-03-01-M	2001			1160	C
HR03-02M	ECO-R	HR-03-02-M	2001			435	C
HR03-03M	ECO-R	HR-03-03-M	2001			164	A
HR04-01B	ECO-R	HR-04-01-M	2001			1240	C
HR04-02M	ECO-R	HR-04-02-M	2001			889	C
HR04-03M	ECO-R	HR-04-03-M	2001			167	A
HR05-01M	ECO-R	HR-05-01-M	2001			1360	C
HR05-02M	ECO-R	HR-05-02-M	2001			460	C
HR05-03M	ECO-R	HR-05-03-M	2001			118	
HR06-01M	ECO-M	HR-06-01-M	2001			1440	C
HR06-02M	ECO-M	HR-06-02-M	2001			1200	C
HR06-03M	ECO-M	HR-06-03-M	2001			1450	C
HR06-04M	ECO-M	HR-06-04-M	2001			433	C
HS1N0003	ECO-R	HS-1N-0003-M	2000			1570	C
HS1N0050	ECO-R	HS-1N-0050-M	2000			1020	C
HS1N0100	ECO-R	HS-1N-0100-M	2000			554	C
HS1N0250	ECO-R	HS-1N-0250-M	2000			281	B

Table CK1. (cont.)

Station	Zone	Sample ID	Event	Copper	Tissue Threshold Concentrations[a]	Zinc		Tissue Threshold Concentrations[a]
				mg/kg	A = 25 - 60	μg/g		A = 150 - 290
				dry	B = 35 - 90	dry		B = 190 - 350
					C = 70 - 110			C = 300 - 400
HS1N1000	ECO-R	HS-1N-1000-M	2000			153		
HS1S0003	ECO-R	HS-1S-0003-M	2000			1500		C
HS1S0050	ECO-R	HS-1S-0050-M	2000			352		C
HS1S0100	ECO-R	HS-1S-0100-M	2000			207		B
HS1S0250	ECO-R	HS-1S-0250-M	2000			148		
HS1S1000	ECO-R	HS-1S-1000-M	2000			111		
HS1S1600	ECO-R	HS-1S-1600-M	2000			96.1		
HS2N0003	ECO-R	HS-2N-0003-M	2000			2750		C
HS2N0050	ECO-R	HS-2N-0050-M	2000			1880		C
HS2N0100	ECO-R	HS-2N-0100-M	2000			1040		C
HS2N0250	ECO-R	HS-2N-0250-M	2000			516		C
HS2N1000	ECO-R	HS-2N-1000-M	2000			237		B
HS2S0003	ECO-R	HS-2S-0003-M	2000			1200		C
HS2S0050	ECO-R	HS-2S-0050-M	2000			321		C
HS2S0100	ECO-R	HS-2S-0100-M	2000			255		B
HS2S0250	ECO-R	HS-2S-0250-M	2000			138		
HS2S1000	ECO-R	HS-2S-1000-M	2000			118		
HS3N0003	ECO-R	HS-3N-0003-M	2000			1180		C
HS3N0050	ECO-R	HS-3N-0050-M	2000			856		C
HS3N0100	ECO-R	HS-3N-0100-M	2000			695		C
HS3N0250	ECO-R	HS-3N-0250-M	2000			259		B
HS3N1000	ECO-R	HS-3N-1000-M	2000			158		A
HS3N1600	ECO-R	HS-3N-1600-M	2000			169		A
HS3S0003	ECO-R	HS-3S-0003-M	2000			2860		C
HS3S0050	ECO-R	HS-3S-0050-M	2000			751		C
HS3S0100	ECO-R	HS-3S-0100-M	2000			453		C
HS3S0250	ECO-R	HS-3S-0250-M	2000			222		B
HS3S1000	ECO-R	HS-3S-1000-M	2000			112		
MI-02M	ECO-M	MI-02-M	2001			589		C
MI-104	ECO-R	MS0024	2003			74.5		
MI-107	ECO-R	MS0020	2003			137		
MI-108	ECO-R	MS0023	2003			386		C
MI-25-M	ECO-R	MI-25-M	2002			440		C
MI-26-M	ECO-R	MI-26-M	2002			166		A
MI-42-M	ECO-M	MI-42-M	2002			611		C
MI-45-M	ECO-M	MI-45-M	2002			748		C
PO-01M	ECO-P	PO-01-M	2001			1370	*J*	C
PO-02M	ECO-P	PO-02-M	2001			2540	*J*	C
PO-04M	ECO-P	PO-04-M	2001			2090	*J*	C
PO-05M	ECO-P	PO-05-M	2001			6480	*J*	C
PO-06M	ECO-P	PO-06-M	2001			3950	*J*	C
PO-07M	ECO-P	PO-07-M	2001			1580	*J*	C
PO-09M	ECO-P	PO-09-M	2001			1560	*J*	C
PO-10M	ECO-P	PO-10-M	2001			1930	*J*	C
PO-11M	ECO-P	PO-11-M	2001			1260	*J*	C
PO-13M	ECO-P	PO-13-M	2001			1580	*J*	C
PO-15M	ECO-P	PO-15-M	2001			1500	*J*	C
PO-16M	ECO-P	PO-16-M	2001			1520	*J*	C
PO-17M	ECO-P	PO-17-M	2001			1550	*J*	C

Table CK1. (cont.)

Station	Zone	Sample ID	Event	Copper	Tissue Threshold Concentrations[a]	Zinc		Tissue Threshold Concentrations[a]
				mg/kg	A = 25 - 60	μg/g		A = 150 - 290
				dry	B = 35 - 90	dry		B = 190 - 350
					C = 70 - 110			C = 300 - 400
PO-18M	ECO-P	PO-18-M	2001			1480	*J*	C
TT1-0100	ECO-P	MS0005	2003	24.2		8120		C
TT1-1000	ECO-P	MS0008	2003	4.56		869		C
TT2-0010	ECO-P	MS0004	2003	21.6		2910		C
TT2-0100	ECO-P	MS0003	2003	13.1		1340		C
TT2-1000	ECO-P	MS0006	2003	3.85		251		B
TT3-0010	ECO-R	MS0002	2003	16.8		1110		C
TT3-0100	ECO-R	MS0001	2003	9.73		595		C
TT3-1000	ECO-R	MS0015	2003	3.49		135		
Reference								
TS-REF-7	ECOREF	MS0011	2003	3.73		47.9		
TS-REF-8	ECOREF	MS0010	2003	4.35		64		
TS-REF10	ECOREF	MS0009	2003	3.29		55		

Note: [a] Tissue threshold concentration ranges defined as follows based on effects thresholds reported for multiple species in Folkeson and Andersson-Bringmark (1988).

A - exceeds minimum threshold for first signs of reduction in cover
B - exceeds minimum threshold for obvious reductions in cover
C - exceeds minimum apparent survival thresholds (some dead individuals observed)

Both site and literature reference samples were unwashed.

J - estimated value

Data Sources: Exponent (2002a)
Ford and Hasselbach (2001)
Exponent (2003c) and Appendix A of this document
Further detail is provided in Appendix Table C-21

Table CK2. Comparison of tissue threshold concentrations in lichen samples

Station	Sample ID	Event	Taxon	Zinc		Tissue Threshold Concentrations[a]
				μg/g dry		A = 480 - 1,300 B = 550 - 1,800 C = 600 - 2,200
Site						
HR01-02L	HR-01-02-L	2001	*Peltigera*	1610		C
HR02-02L	HR-02-02-L	2001	*Peltigera*	545	*J*	A
HR02-03L	HR-02-03-L	2001	*Peltigera*	82.2	*J*	
HR03-03L	HR-03-03-L	2001	*Peltigera*	115	*J*	
HR05-03L	HR-05-03-L	2001	*Peltigera*	85.2	*J*	
HR07-01B	HR-07-01-L	2001	*Peltigera*	1720	*J*	C
HR07-02L	HR-07-02-L	2001	*Peltigera*	1040	*J*	C
HR07-03L	HR-07-03-L	2001	*Peltigera*	185	*J*	
HR07-04L	HR-07-04-L	2001	*Peltigera*	121	*J*	
PO-04L	PO-04-L	2001	*Peltigera*	1010	*J*	C
PO-11L	PO-11-L	2001	*Peltigera*	1020	*J*	C
PO-17L	PO-17-L	2001	*Peltigera*	1050	*J*	C
TT2-0010	LI0018	2004	*Peltigera*	780		C
TT2-0100	LI0008	2004	*Peltigera*	292		
TT2-1000	LI0007	2004	*Peltigera*	137		
TT3-0010	LI0010	2004	*Peltigera*	209		
TT3-0100	LI0037	2004	*Peltigera*	119	*J*	
TT3-1000	LI0016	2004	*Cladina*	81.9		
TT3-1000	LI0017	2004	*Peltigera*	94.4		
TT5-0010	LI0038	2004	*Peltigera*	594		B
TT5-0100	LI0006	2004	*Peltigera*	572		B
TT5-1000	LI0002	2004	*Peltigera*	531		A
TT5-2000	LI0019	2004	*Cladina*	278		
TT6-0010	LI0034-D	2004	*Peltigera*	351	*J*	
TT6-0010	LI0036	2004	*Cladina*	317	*J*	
TT6-0100	LI0022	2004	*Cladina*	420	*J*	
TT6-0100	LI0023	2004	*Peltigera*	392	*J*	
TT6-1000	LI0020	2004	*Peltigera*	335	*J*	
TT6-1000	LI0021	2004	*Cladina*	386	*J*	
TT6-2000	LI0026	2004	*Peltigera*	163	*J*	
TT6-2000	LI0027	2004	*Cladina*	141	*J*	
TT7-0010	LI0025	2004	*Cladina*	2740	*J*	C
TT7-1000	LI0024	2004	*Cladina*	996	*J*	C
TT7-2000	LI0039	2004	*Cladina*	1260		C
TT8-0010	LI0015	2004	*Peltigera*	627		C
TT8-0100	LI0014	2004	*Peltigera*	397		
TT8-1000	LI0011	2004	*Cladina*	70		
TT8-1000	LI0012-D	2004	*Peltigera*	149		
Reference						
TS-REF-5	LI0028	2004	*Cladina*	45.2		
TS-REF-5	LI0029	2004	*Peltigera*	48.5		
TS-REF-7	LI0030	2004	*Cladina*	26.9		
TS-REF-7	LI0031	2004	*Peltigera*	39.2		
TS-REF11	LI0032	2004	*Cladina*	19.4	*J*	
TS-REF11	LI0033	2004	*Peltigera*	29.7	*J*	

Notes on following page

Table CK2. (cont.)

Note: [a] Tissue threshold concentration ranges defined as follows based on effects thresholds reported for multiple species in Folkeson and Andersson-Bringmark (1988).

A - exceeds minimum threshold for first signs of reduction in cover

B - exceeds minimum threshold for obvious reductions in cover

C - exceeds minimum apparent survival thresholds (some dead individuals observed)

Both site and literature reference samples were unwashed.

J - estimated value

Data Sources: Exponent (2004a) and Appendix E of this document.
Data are presented in Appendix Table G-19.

Table CK3. Food-web exposure modeling results for willow ptarmigan

		NOAEL Hazard Quotient		LOAEL Hazard Quotient	
Assessment Unit	Chemical	Mean	95% UCL	Mean	95% UCL
Port	Lead	2.4	6.2	0.84	2.2
Port	Mercury	0.40	1.2	0.20	0.62
Port	Zinc (TRV2)	0.82	1.3	0.48	0.74
Road	Barium	1.2	1.7	0.59	0.87
Mine	Barium	1.9	4.0	0.94	2.0
Mine	Lead	1.6	3.5	0.55	1.2
Mine	Zinc (TRV2)	0.51	1.4	0.29	0.81

Note: Results shown only for chemicals with NOAEL-based hazard quotients >1.0.

For 10 CoPCs (aluminum, antimony, arsenic, cadmium, chromium, cobalt, molybdenum, selenium, thallium, and vanadium) all hazard quotients were less than 1.0.
No hazard quotients were exceeded for the reference area; all values were < 1.0.

95% UCL - 95 percent upper confidence limit on the mean
LOAEL - lowest-observed-adverse-effect level
NOAEL - no-observed-adverse-effect level

Table CS1. Comparison of juvenile Dolly Varden tissue concentrations with effects thresholds

	Source[a]	Date Collected	N	Total Cadmium Min.	Total Cadmium Max.	Total Lead Min.	Total Lead Max.	Total Selenium Min.	Total Selenium Max.	Total Zinc Min.	Total Zinc Max.
Anxiety Ridge Creek (all)	ADFG	1993–2002	61	0.017	0.308	0.001	0.612	0.010	2.01	11.48	36.12
ARC at Haul Road	ADFG	1993–2000	31	0.022	0.090	0.041	0.612	0.529	1.37	--	--
ARC Upstream	ADFG	2002	15	0.017	0.224	0.001	0.101	0.010	2.01	11.48	36.12
ARC Downstream	ADFG	2002	15	0.039	0.308	0.031	0.138	0.895	2.01	21.97	32.56
Literature values[b] for tissue residue and effect (ppm)											
No effects (range)[c]				0.036–5.0		0.34–5.1		0.12–19		4.5–480	
No effects (range)[d]				0.04–2		0.34–5.1		0.2–0.8		4.5–60	
Effects (range)[c]				0.12–8.0		0.4–4.0		0.66–4.6		40–60	
Effects (range)[d]				0.12–4.0		0.4–4.0		0.66–2.08		--	

Note: Concentrations are reported in ppm wet wt (converted from dry wt).
Based on studies with ecologically relevant endpoints (survival, growth, or reproduction).
If multiple effects thresholds were provided in a single study, the highest no effects threshold value was used.
If multiple effects thresholds were provided in a single study, the lowest effects threshold value was used.
ADFG - Alaska Department of Fish and Game
ARC - Anxiety Ridge Creek
-- - Not available

[a] 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.

[b] 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.

[c] Ranges of whole body tissue concentrations for all freshwater fish species (Atlantic salmon, bluegill, brook trout, Chinook salmon, dace, fathead minnow, flagfish, guppy, largemouth bass, perch, rainbow trout, stickleback) exposed to chemicals in water or their diet for at least 30 days.

[d] Ranges of whole body tissue concentrations for only freshwater salmonids (Altantic salmon, brook trout, Chinook salmon, rainbow trout) exposed to chemicals in water or their diet for at least 30 days.

Table WH1. DMTS haul truck spill sites summary information

Spill Site	Date of Spill	Spill Type	Tons Spilled	DMTS Mile Post	Grid Reference Monument Location: Monument Site	Latitude (North)	Longitude (West)
SP-01	01/12/90	Zinc	15	41.85	SP01-001	67.94109	163.05006
SP-02	01/17/90	Zinc	72	48.1	SP02-001	68.01279	162.93477
SP-03	08/02/90	Zinc	36	4.1	SP03-019	67.60539	163.94122
SP-04	09/03/90	Zinc	35	29.4	SP04-001	67.82995	163.34785
SP-05	09/18/90	Zinc	36	4.95	SP05-001	67.61495	163.92287
SP-06	12/01/91	Lead	30	40.3	SP06-001	67.93362	163.09655
SP-07	02/20/92	Lead	72	8.5	SP07-002	67.63837	163.80674
SP-08	03/20/92	Lead	15	21.1	SP08-010	67.76672	163.56600
SP-09	07/29/92	Zinc	37	48.85	SP09-001	68.02160	162.93450
SP-10	07/14/93	Zinc	35	51.3	SP10-001	68.04481	162.86582
SP-11	12/15/93	Zinc	28	26.65	SP11-001	67.80349	163.42157
SP-12	09/06/94	Zinc	36	48.75	SP12-001/007	68.01966	162.93574
SP-13	08/05/96	Zinc	35	32.3	SP13-001	67.86341	163.29792
SP-14	12/10/96	Zinc	37	48.65	SP14-014	68.01856	162.93419
SP-15	01/02/97	Zinc	17	27	SP15-002	67.80696	163.41252
SP-16	08/19/97	Zinc	15	51.05	SP16/26-001	68.04253	162.87135
SP-17	08/21/97	Zinc	10	1	SP17-001	67.58687	164.02718
SP-18	01/17/98	Zinc	17	35	SP18-001	67.89438	163.24367
SP-19	02/07/98	Zinc	45	27.25	SP19-001	67.80876	163.40421
SP-20	04/17/98	Zinc	0.4	32.6	SP20-001	67.86890	163.29753
SP-21	07/11/98	Zinc	20	42.4	SP21-001	67.93950	163.07366
SP-22	08/01/98	Lead	76	RT	SP22-001	67.94109	163.05007
SP-23	11/21/98	Zinc	40	41.75	SP23-001	67.94623	163.04388
SP-24	01/06/99	Zinc	72.5	45	SP24-001	67.97663	162.98434
SP-25	01/21/99	Lead	38	9.02	SP25-008	67.64293	163.79299
SP-26	07/19/99	Lead	66	51.05	SP16/26-001	68.04253	162.87135
SP-27	10/09/00	Lead	30	32.5	Station "4"	67.92447	163.10250
SP-28	12/22/00	Zinc	40	44.7	Station "3"	67.97392	162.99184
SP-29	02/16/01	Zinc	14	42.2	Station "104"	67.94592	163.04355
SP-30	07/20/01	Zinc	10	39.25	Station "102"	67.92447	163.10250
SP-31A	03/22/98	Zinc	1 [a]	48-53	SP31A-001	68.01159	162.93434
SP-31B	03/22/98	Zinc	1 [a]	48-53	Road Side	68.01533	162.93524
SP-31C	03/22/98	Zinc	1 [a]	48-53	Road Side	68.01619	162.93483

Source: Teck Cominco (2003c)

[a] Total tonnage spilled at site SP-31 was estimated at 1 ton, distributed among three subsites.

Table WH2. Concentrate truck spill evaluation summary

Spill ID	Date of Spill	Closeout Date of Spill Re-evaluation	Re-evaluation Document
SP-01	January 12, 1990	May 2005	Close out letter from TCAK to ADEC, Div. of Spill Prevention and Response[1]
SP-02	January 17, 1990	May 2005	Close out letter from TCAK to ADEC, Div. of Spill Prevention and Response[1]
SP-03	August 09, 1990	May 2003	Concentrate Spill Site Recovery and Restoration Report for SP-03[2]
SP-04	September 03, 1990	May 2005	Close out letter from TCAK to ADEC, Div. of Spill Prevention and Response[1]
SP-05	September 18, 1990	May 2003	Concentrate Spill Site Recovery and Restoration Report for SP-05[3]
SP-06	December 1, 1991	May 2005	Close out letter from TCAK to ADEC, Div. of Spill Prevention and Response[1]
SP-07	February 20, 1992	February 2003	Report on the 2002 Spill Site Characterization Sampling Program[4]
SP-08	March 20, 1992	May 2005	Close out letter from TCAK to ADEC, Div. of Spill Prevention and Response[1]
SP-09	July 29, 1992	May 2005	Close out letter from TCAK to ADEC, Div. of Spill Prevention and Response[1]
SP-10	July 14, 1993	February 2004	2002–2003 DMTS Concentrate Spill Site Characterization Report for SP-10[5]
SP-11	December 16, 1993	May 2005	Close out letter from TCAK to ADEC, Div. of Spill Prevention and Response[1]
SP-12	September 06, 1994	February 2003	Report on the 2002 Spill Site Characterization Sampling Program[4]
SP-13	August 05, 1996	February 2004	2002–2003 DMTS Concentrate Spill Site Characterization Report for SP-13[6]
SP-14	December 10, 1996	February 2003	Report on the 2002 Spill Site Characterization Sampling Program[4]
SP-15	January 02, 1997	February 2004	2002–2003 DMTS Concentrate Spill Site Characterization Report for SP-15[7]
SP-16	August 19, 1997	May 2005	Close out letter from TCAK to ADEC, Div. of Spill Prevention and Response[1]
SP-17	August 21, 1997	February 2004	2002–2003 DMTS Concentrate Spill Site Characterization Report for SP-17[8]
SP-18	January 17, 1998	May 2005	Close out letter from TCAK to ADEC, Div. of Spill Prevention and Response[1]
SP-19	February 7, 1998	May 2005	Close out letter from TCAK to ADEC, Div. of Spill Prevention and Response[1]
SP-20	April 17, 1998	February 2003	Report on the 2002 Spill Site Characterization Sampling Program[4]

Spill ID	Date of Spill	Closeout Date of Spill Re-evaluation	Re-evaluation Document
SP-21	July 11, 1998	May 2005	Close out letter from TCAK to ADEC, Div. of Spill Prevention and Response[1]
SP-22	August 1, 1998	May 2005	Close out letter from TCAK to ADEC, Div. of Spill Prevention and Response[1]
SP-23	November 21, 1998	February 2003	Report on the 2002 Spill Site Characterization Sampling Program[4]
SP-24	January 6, 1999	May 2005	Close out letter from TCAK to ADEC, Div. of Spill Prevention and Response[1]
SP-25	January 21, 1999	May 2005	Close out letter from TCAK to ADEC, Div. of Spill Prevention and Response[1]
SP-26	July 19, 1997	May 2005	Close out letter from TCAK to ADEC, Div. of Spill Prevention and Response[1]
SP-27	October 09, 2000	February 2004	2002–2003 DMTS Concentrate Spill Site Characterization Report for SP-27[9]
SP-28	December 28, 2000	May 2005	Close out letter from TCAK to ADEC, Div. of Spill Prevention and Response[1]
SP-29	February 16, 2001	February 2004	2002–2003 DMTS Concentrate Spill Site Characterization Report for SP-29[10]
SP-30	July 20, 2001	February 2004	2002–2003 DMTS Concentrate Spill Site Characterization Report for SP-30[11]
SP-31	March 22, 1998	February 2003	Report on the 2002 Spill Site Characterization Sampling Program[4]

[1] Teck Cominco. 2005. 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.

[2] Teck Cominco. 2003a. 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.

[3] Teck Cominco. 2003b. 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.

[4] Teck Cominco. 2003c. 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.

[5]Teck Cominco. 2004a. 2002–2003 DMTS concentrate spill site characterization report, spill site SP-10. Teck Cominco Alaska Incorporated, Anchorage, AK.

[6]Teck Cominco. 2004b. 2002–2003 DMTS concentrate spill site characterization report, spill site SP-13. Teck Cominco Alaska Incorporated, Anchorage, AK.

[7]Teck Cominco. 2004c. 2002–2003 DMTS concentrate spill site characterization report, spill site SP-15. Teck Cominco Alaska Incorporated, Anchorage, AK.

[8]Teck Cominco. 2004d. 2002 DMTS concentrate spill site characterization report, spill site SP-17. Teck Cominco Alaska Incorporated, Anchorage, AK.

[9]Teck Cominco. 2004e. 2002–2003 DMTS concentrate spill site characterization report, spill site SP-27. Teck Cominco Alaska Incorporated, Anchorage, AK.

[10]Teck Cominco. 2004f. 2002–2003 DMTS concentrate spill site characterization report, spill site SP-29. Teck Cominco Alaska Incorporated, Anchorage, AK.

[11]Teck Cominco. 2004g. 2002–2003 DMTS concentrate spill site characterization report, spill site SP-30. Teck Cominco Alaska Incorporated, Anchorage, AK.

Table JS5a. Locations and receptors for which NOAEL or LOAEL hazard quotients exceed 1.0

Assessment Unit Location	Aluminum	Antimony	Arsenic (arsenate)	Arsenic (arsenite)	Barium	Cadmium	Chromium	Cobalt	Lead	Mercury	Molybdenum	Selenium	Thallium	Vanadium	Zinc
DMTS Road and Port Operations															
Site Stations															
Whole Site	Moose, caribou				Caribou										
Port Site	Moose, fox, caribou				Caribou				Ptarmigan	Ptarmigan					Ptarmigan
Near Mine	Moose, caribou				Ptarmigan, caribou				Ptarmigan, caribou						Ptarmigan
Road Site	Moose, fox, caribou				Ptarmigan, caribou					Owl, fox					
Reference Stations															
Reference Site	Moose, fox, caribou														
Lagoon Environment															
Site Stations															
Control Lagoon	Moose, muskrat														
North Lagoon	Moose, muskrat														
Port Lagoon North	Moose, muskrat								Plover						
Reference Stations															
Reference Lagoon	Moose, muskrat														
Tundra Pond Environment															
Site Stations															
TP1-0100	Muskrat														
TP1-1000	Muskrat							Muskrat							
TP3	Muskrat				Muskrat										
TP4	Muskrat				Muskrat										
Reference Stations															
TP-REF-2	Muskrat														
TP-REF-3	Teal, muskrat			Muskrat	Muskrat		Teal, muskrat							Muskrat	
TP-REF-5	Teal, muskrat		Muskrat	Muskrat	Muskrat		Teal							Muskrat	
Stream Environment															
Site Stations															
ARC-R	Moose, muskrat				Moose, muskrat										
OR-R	Moose, muskrat			Muskrat	Muskrat									Muskrat	
AC-R	Moose														
Reference Stations															
ST-REF-3	Moose, muskrat			Muskrat											
ST-REF-5	Moose, muskrat				Muskrat										
ST-REF-6	Moose, muskrat				Muskrat										
Terrestrial Environment															
Site Stations															
TT2-0010	Vole, shrew, snipe			Shrew	Vole, shrew	Shrew			Shrew	Shrew				Vole, shrew	Shrew
TT2-0100	Vole, shrew				Vole, shrew	Shrew				Shrew		Shrew		Shrew	
TT2-1000	Vole, shrew									Shrew		Shrew			
TT3-0010	Vole, shrew, snipe			Shrew	Vole, shrew	Shrew				Shrew				Vole, shrew	
TT3-0100	Vole, shrew				Vole, shrew	Shrew				Shrew					
TT3-1000	Vole, shrew				Vole										
TT5-0010	Snipe, vole, shrew			Shrew	Vole, shrew	Shrew			Snipe, vole, shrew	Shrew		Shrew		Shrew	Shrew

Table JS5a. (cont.)

Assessment Unit Location	Aluminum	Antimony	Arsenic (arsenate)	Arsenic (arsenite)	Barium	Cadmium	Chromium	Cobalt	Lead	Mercury	Molybdenum	Selenium	Thallium	Vanadium	Zinc
Terrestrial Environment (cont.)															
Site Stations (cont.)															
TT5-0100	Vole, shrew			Shrew	Vole, shrew	Shrew			Snipe, vole, shrew	Shrew				Shrew	Shrew
TT5-1000	Vole, shrew				Vole					Shrew		Shrew			
TT5-2000	Vole, shrew					Shrew				Shrew		Shrew			Shrew
TT6-0010	Vole, shrew, snipe			Vole, shrew	Vole, shrew, snipe	Shrew								Vole, shrew	
TT6-0100	Vole, shrew				Vole, shrew, snipe	Shrew				Shrew					
TT6-1000	Vole, shrew				Vole, shrew	Shrew					Shrew			Shrew	
TT6-2000	Vole				Vole										
TT7-0010	Vole			Vole	Vole				Vole					Vole	
TT7-1000	Vole				Vole				Vole		Vole				
TT7-2000	Vole				Vole										
TT8-0010	Vole				Vole									Vole	
TT8-0100	Vole				Vole										
TT8-1000	Vole														
Reference Stations															
TS-REF-5	Vole, shrew, snipe				Vole, shrew							Shrew		Shrew	
TS-REF-7	Vole				Vole										
TS-REF-11	Vole														

Source: Appendix K tables of this report.

Note:

-0010, -0100, -1000	-	approximate distance of station from DMTS Road or facilities in meters
AC-R	-	Aufeis Creek station, just downstream of the DMTS road crossing
ARC-R	-	Anxiety Ridge Creek station, just downstream of the DMTS road crossing
DMTS	-	DeLong Mountain Regional Transportation System
LOAEL	-	lowest-observed-adverse-effect level
NOAEL	-	no-observed-adverse-effect level
OR-R	-	Omikviorok River station, just downstream of the DMTS road crossing
REF	-	reference stations
ST	-	stream station
TP	-	tundra pond station
TS	-	tundra soil station
TT	-	terrestrial transect station

Table JS5b. Locations and receptors for which only LOAEL hazard quotients exceed 1.0

Assessment Unit Location	Aluminum	Antimony	Arsenic (arsenate)	Arsenic (arsenite)	Barium	Cadmium	Chromium	Cobalt	Lead	Mercury	Molybdenum	Selenium	Thallium	Vanadium	Zinc
DMTS Road and Port Operations															
Site Stations															
Whole Site	Caribou				Caribou										
Port Site	Caribou, fox								Ptarmigan						
Near Mine	Caribou				Ptarmigan, caribou				Ptarmigan						
Road Site	Caribou									Fox, owl					
Reference Stations															
Reference Site	Caribou														
Lagoon Environment															
Site Stations															
Control Lagoon															
North Lagoon															
Port Lagoon North															
Reference Stations															
Reference Lagoon															
Tundra Pond Environment															
Site Stations															
TP1-0100															
TP1-1000															
TP3															
TP4					Muskrat										
Reference Stations															
TP-REF-2															
TP-REF-3	Muskrat														
TP-REF-5	Muskrat														
Stream Environment															
Site Stations															
ARC-R	Muskrat														
OR-R	Muskrat														
AC-R															
Reference Stations															
ST-REF-3	Muskrat														
ST-REF-5	Muskrat														
ST-REF-6	Muskrat														

Table JS5b. (cont.)

Assessment Unit Location	Aluminum	Antimony	Arsenic (arsenate)	Arsenic (arsenite)	Barium	Cadmium	Chromium	Cobalt	Lead	Mercury	Molybdenum	Selenium	Thallium	Vanadium	Zinc
Terrestrial Environment															
Site Stations															
TT2-0010	Vole, shrew				Vole, shrew										
TT2-0100	Vole, shrew														
TT2-1000															
TT3-0010	Vole, shrew				Vole, shrew										
TT3-0100	Vole, shrew				Vole, shrew										
TT3-1000															
TT5-0010	Vole, shrew				Vole, shrew										
TT5-0100	Vole, shrew				Vole, shrew										
TT5-1000															
TT5-2000															
TT6-0010	Vole, shrew				Vole, shrew										
TT6-0100	Vole, shrew				Vole, shrew										
TT6-1000	Vole				Shrew										
TT6-2000															
TT7-0010	Vole				Vole										
TT7-1000	Vole				Vole										
TT7-2000					Vole										
TT8-0010	Vole				Vole										
TT8-0100	Vole				Vole										
TT8-1000															
Reference Stations															
TS-REF-5 Site	Vole, shrew														
TS-REF-7 Site															
TS-REF-11 Site															

Source: Appendix K tables of this report.

Note:

-0010, -0100, -1000	- approximate distance of station from DMTS Road or facilities in meters		REF	- reference stations
AC-R	- Aufeis Creek station, just downstream of the DMTS road crossing		ST	- stream station
ARC-R	- Anxiety Ridge Creek station, just downstream of the DMTS road crossing		TP	- tundra pond station
DMTS	- DeLong Mountain Regional Transportation System		TS	- tundra soil station
LOAEL	- lowest-observed-adverse-effect level		TT	- terrestrial transect station
OR-R	- Omikviorok River station, just downstream of the DMTS road crossing			

Table JS6. Summary of LOAEL hazard quotient exceedances

	Aluminum	Antimony	Arsenic (arsenate)	Arsenic (arsenite)	Barium	Cadmium	Chromium	Cobalt	Lead	Mercury	Molybdenum	Selenium	Thallium	Vanadium	Zinc
Tundra vole															
Site stations	13/20	--	0/20	0/20	12/20	0/20	0/20	0/20	0/20	0/20	0/20	0/20	0/20	0/20	0/20
Reference stations	1/3	--	0/3	0/3	0/3	0/3	0/3	0/3	0/3	0/3	0/3	0/3	0/3	0/3	0/3
Common snipe															
Site stations	--	--	0/13	0/13	0/13	0/16	0/13	--	0/16	0/16	0/13	0/13	0/13	--	0/16
Reference stations	--	--	0/2	0/2	0/2	0/3	0/2	--	0/3	0/3	0/2	0/2	0/2	--	0/3
Lapland longspur															
Site stations	--	--	0/13	0/13	0/13	0/13	0/13	--	0/13	0/13	0/13	0/13	0/13	--	0/13
Reference stations	--	--	0/1	0/1	0/1	0/1	0/1	--	0/1	0/1	0/1	0/1	0/1	--	0/1
Black-bellied plover															
Site stations	--	--	0/3	0/3	0/3	--	--	--	0/3	0/3	0/3	0/3	0/3	--	0/3
Reference stations	--	--	0/1	0/1	0/1	--	--	--	0/1	0/1	0/1	0/1	0/1	--	0/1
Green-winged teal															
Site stations	--	--	0/6	0/6	0/6	0/6	0/6	--	0/6	0/6	0/6	0/6	0/6	--	0/6
Reference stations	--	--	0/6	0/6	0/6	0/6	0/6	--	0/6	0/6	0/6	0/6	0/6	--	0/6
Snowy owl															
Site stations	--	--	0/2	0/2	0/2	0/2	0/2	--	0/2	1/2	0/2	0/2	0/2	--	0/2
Reference stations	--	--	0/1	0/1	0/1	0/1	0/1	--	0/1	0/1	0/1	0/1	0/1	--	0/1
Willow ptarmigan															
Site stations	--	--	0/3	0/3	1/3	0/3	0/3	--	2/3	0/3	0/3	0/3	0/3	--	0/3
Reference stations	--	--	0/1	0/1	0/1	0/1	0/1	--	0/1	0/1	0/1	0/1	0/1	--	0/1
Brant															
Site stations	--	--	0/3	0/3	0/3	0/3	0/3	--	0/3	0/3	0/3	0/3	0/3	--	0/3
Reference stations	--	--	0/1	0/1	0/1	0/1	0/1	--	0/1	0/1	0/1	0/1	0/1	--	0/1
Arctic fox															
Site stations	1/2	--	0/2	0/2	0/2	0/2	0/2	0/2	0/2	1/2	0/2	0/2	0/2	0/2	0/2
Reference stations	0/1	--	0/1	0/1	0/1	0/1	0/1	0/1	0/1	0/1	0/1	0/1	0/1	0/1	0/1
Caribou															
Site stations	4/4	--	0/4	0/4	2/4	0/4	0/4	0/4	0/4	0/4	0/4	0/4	0/4	0/4	0/4
Reference stations	1/1	--	0/1	0/1	0/1	0/1	0/1	0/1	0/1	0/1	0/1	0/1	0/1	0/1	0/1
Moose															
Site stations	0/10	--	0/10	0/10	0/10	0/10	0/10	0/10	0/10	0/10	0/10	0/10	0/10	0/10	0/10
Reference stations	0/5	--	0/5	0/5	0/5	0/5	0/5	0/5	0/5	0/5	0/5	0/5	0/5	0/5	0/5
Tundra shrew															
Site stations	8/13	--	0/13	0/13	8/13	0/13	0/13	0/13	0/13	0/13	0/13	0/13	0/13	0/13	0/13
Reference stations	1/1	--	0/1	0/1	0/1	0/1	0/1	0/1	0/1	0/1	0/1	0/1	0/1	0/1	0/1
Muskrat															
Site stations	2/9	--	0/9	0/9	1/9	0/9	0/9	0/9	0/9	0/9	0/9	0/9	0/9	0/9	0/9
Reference stations	5/7	--	0/7	0/7	0/7	0/7	0/7	0/7	0/7	0/7	0/7	0/7	0/7	0/7	0/7

Source: Appendix K tables of this report.

Note: Ratios represent number of LOAEL exceedances/number of sites evaluated.
Shaded cells are those with one or more exceedances.

This summary is based on the most conservative scenarios presented in Appendix K.

-- - analyte not analyzed

LOAEL - lowest-observed-adverse-effect level

Table JS7. Summary of observed and predicted ecological effects[a]

Terrestrial Habitats	Observed or Predicted Effects		
Receptor	Near Port	Near Mine[b]	DMTS Road
Caribou	--	--	--
Moose	--	--	--
Lapland longspur	--	--	--
Snowy owl	--	--	--
Arctic fox	--	--	--
Ptarmigan	yes[c]	yes[c]	--
Tundra vole	--	--	--
Tundra shrew	--	--	--
Vegetation	yes[d]	yes[b,e]	yes[d]

Freshwater Habitats	Observed or Predicted Effects			
Receptor	Aufeis Creek	Omikiviorok Creek	Anxiety Ridge Creek	Tundra Ponds
Benthic macroinvertebrates	--	--	--	[f]
Fish	--	--	--[g]	--[h]
Green-winged teal	--	--	--	--
Muskrat	--	--	--	--
Moose	--	--	--	--
Common snipe	--	--	--	--
Vegetation	[f]	[f]	[f]	--[i]

Coastal Lagoon Habitats	Observed or Predicted Effects
Receptor	Lagoons[j]
Benthic macroinvertebrates	--
Fish	--[k]
Brant	--
Muskrat	--
Moose	--
Black-bellied plover	--
Vegetation	--

Source: Summary based on Tables 6-42 and 6-43, and the interpretation of ecological significance (Section 6.7).

Note: -- - indicates very low or no likelihood of adverse effects

[a] Observed or predicted effects indicated as "yes" are to be addressed in a risk management plan, as discussed in Section 8.

[b] The areas evaluated near the mine were outside the mine boundary. The area within the mine boundary was beyond the scope of this assessment.

[c] Potential for adverse effects from lead.

[d] Vegetation survey parameters were statistically compared to reference area data (Tables 6-3 and 6-37), and several differences were observed, as summarized in Table 6-37. No individual metals were isolated as primary causative factors. Multiple causative factors are likely.

[e] The hillslope community vegetation did not show significant difference from the reference site (Tables 6-3 and 6-37). However, at one transect station just west of the mine's ambient air/solid waste permit boundary, some shrubs appeared to be in poor condition.

[f] Not evaluated.

[g] Cadmium and lead levels in some juvenile Dolly Varden exceeded conservative screening levels for fish tissue, but were also within the range of no-effects levels (Table 6-27).

[i] Exception: Effects possible from lead and zinc in ephemeral tundra ponds located within 100 m of port facility structures, based on exceedances of literature-derived effects thresholds. However, tundra pond vegetation appeared healthy during field sampling.

[j] Lagoons located within the port site boundary.

[k] No fish were present in port site lagoons, as they have no open water connections to the Chukchi Sea.

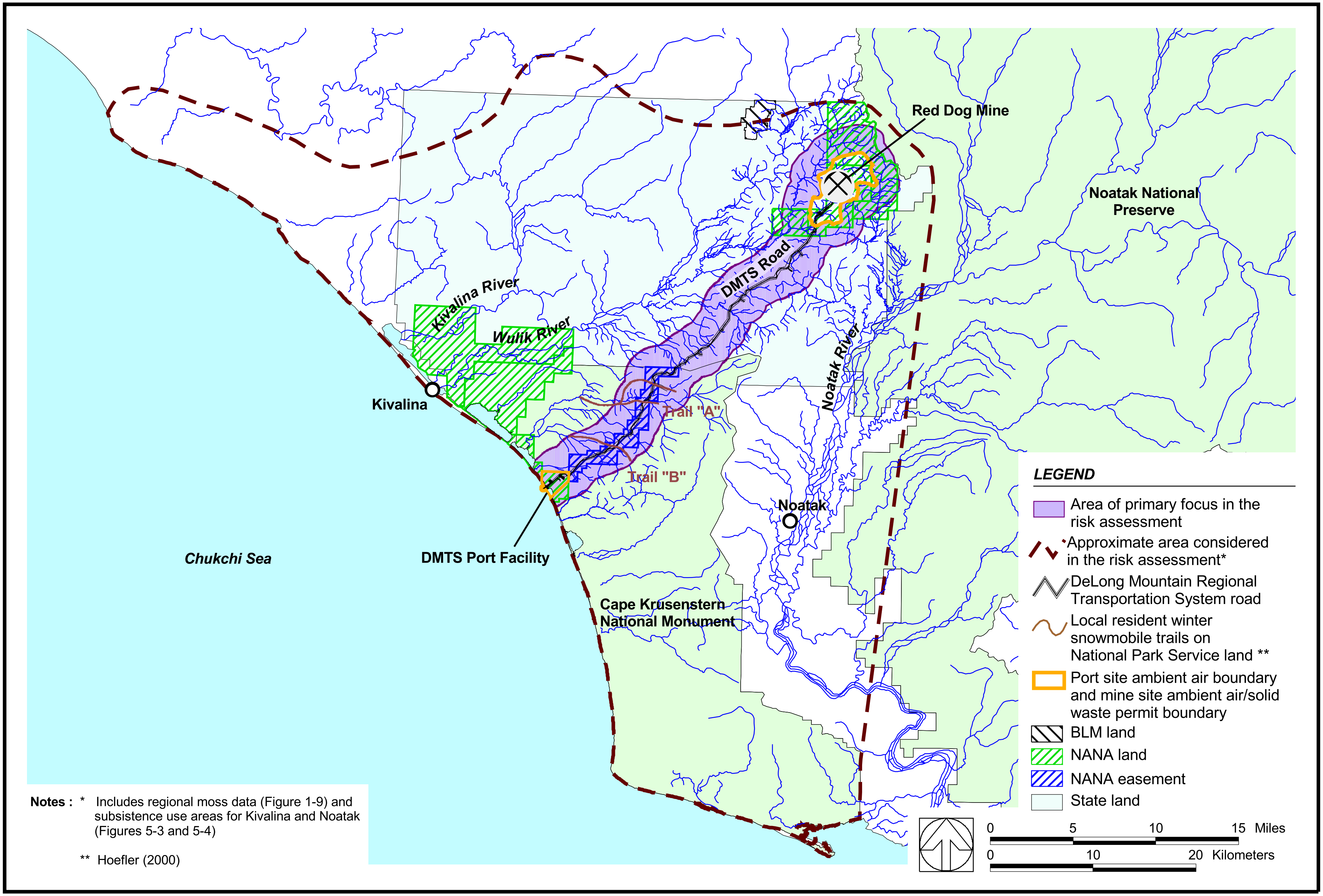
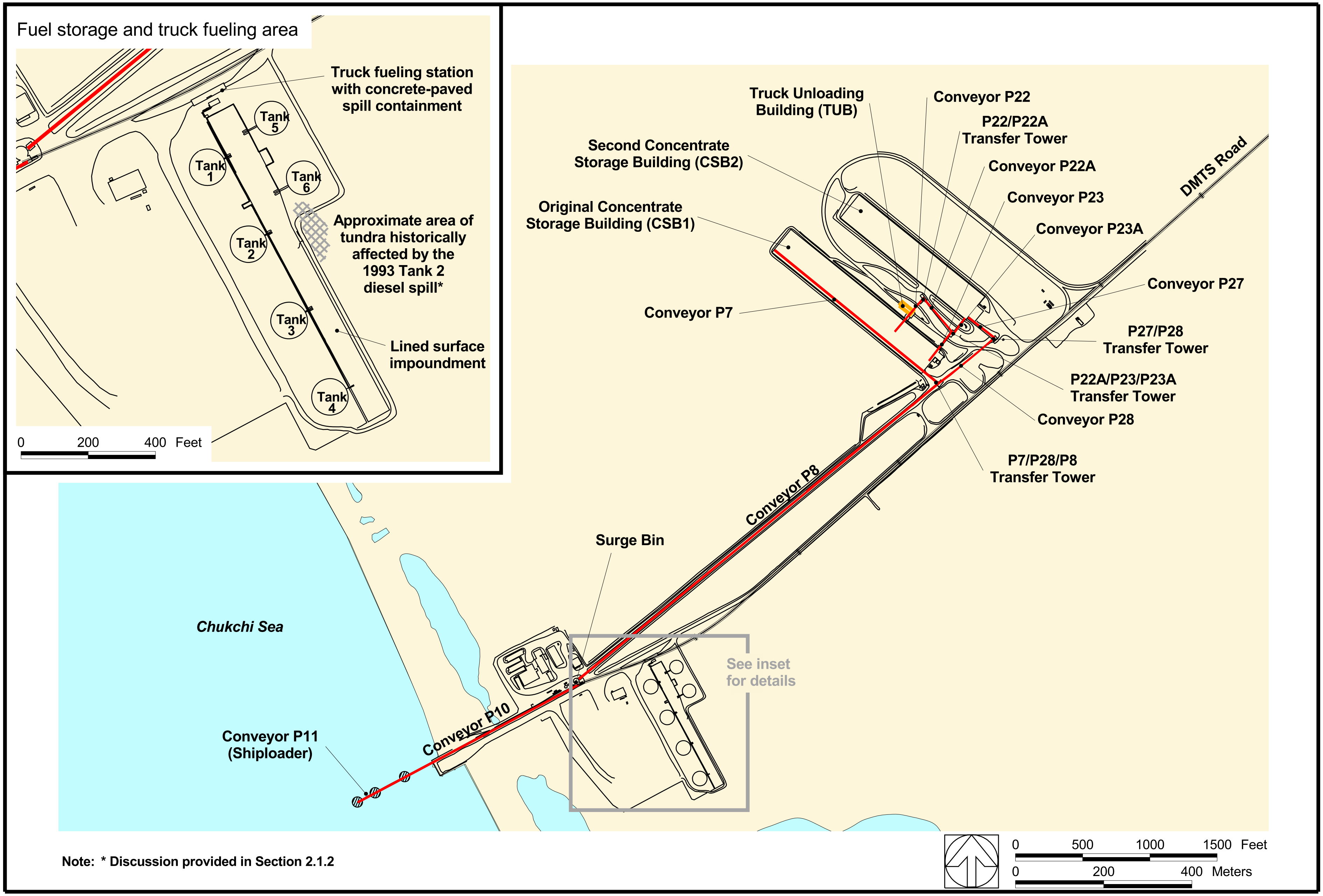


Figure 1-7. Areas evaluated in the risk assessment

Exponent

8601997.007 5400 | Jun 24, 2007 | Fig 1-7 evaluation view | Fig 1-7 RA evaluation layout | j:\red_dog\projects\post_fsp_2003_base_figures.apr



Note: * Discussion provided in Section 2.1.2

Figure 1-8. Port site storage and conveyance features map

Exponent®

8601997.001 3400/3500 | Nov 12, 2004 | Fig 1-8 tank detail view | Fig 1-8 Port detail layout | j:\red_dog\projects\port_conveyance.apr

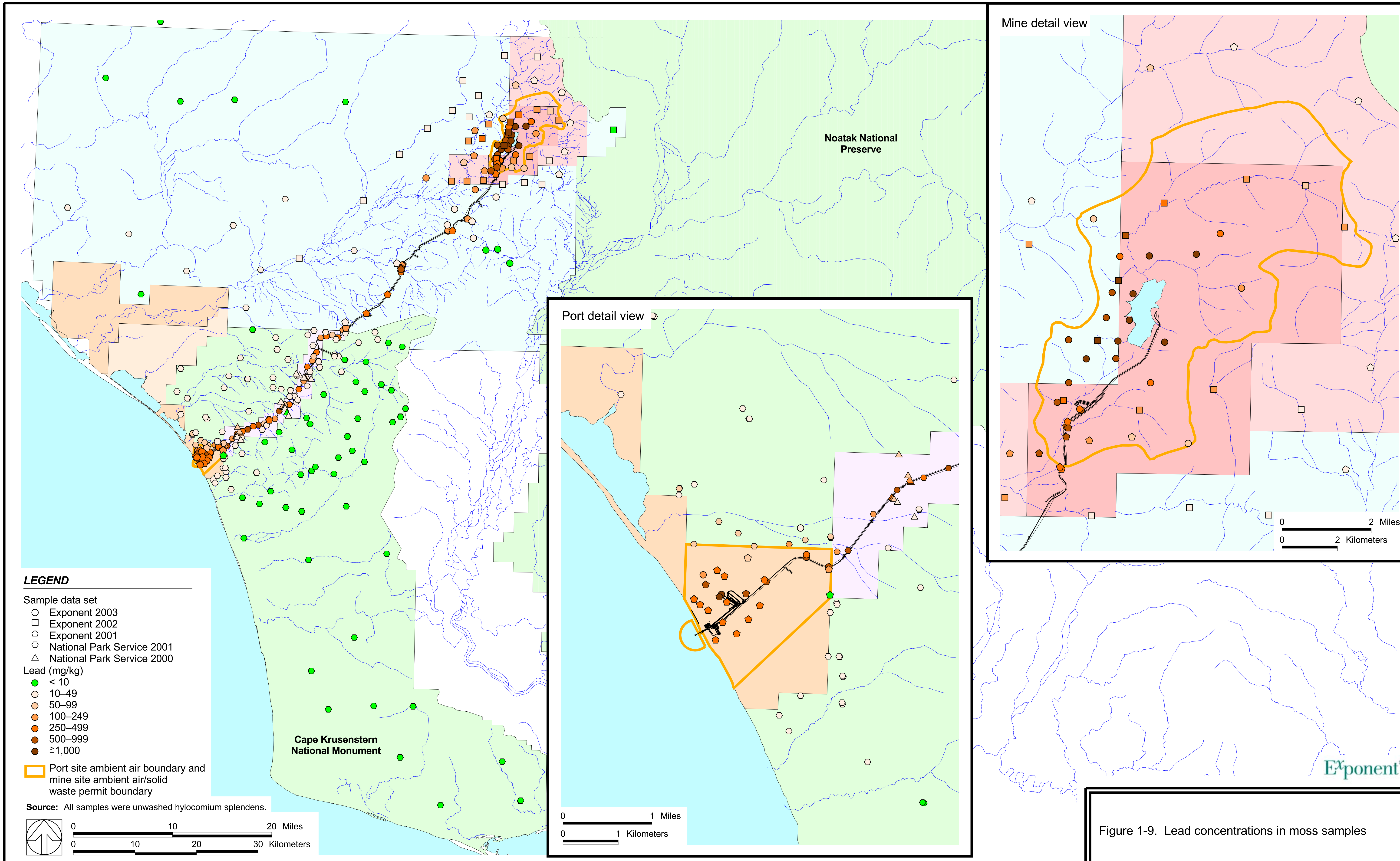


Figure 1-9. Lead concentrations in moss samples

8601997.001 4400 | Jan 5, 2006 | all moss view | all moss 11x17 layout | j:\red_dog\projects\moss0106.apr

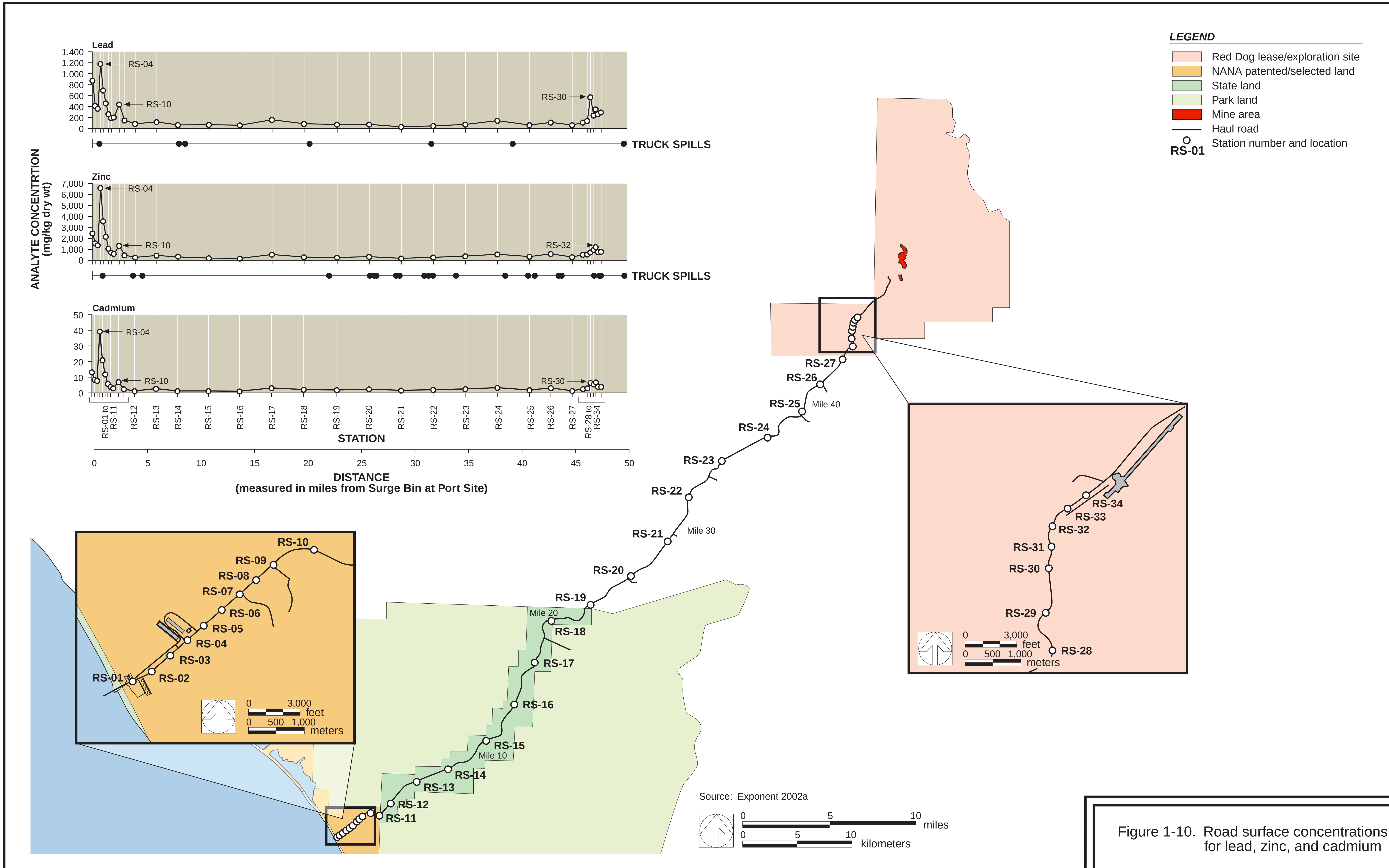


Figure 1-10. Road surface concentrations for lead, zinc, and cadmium

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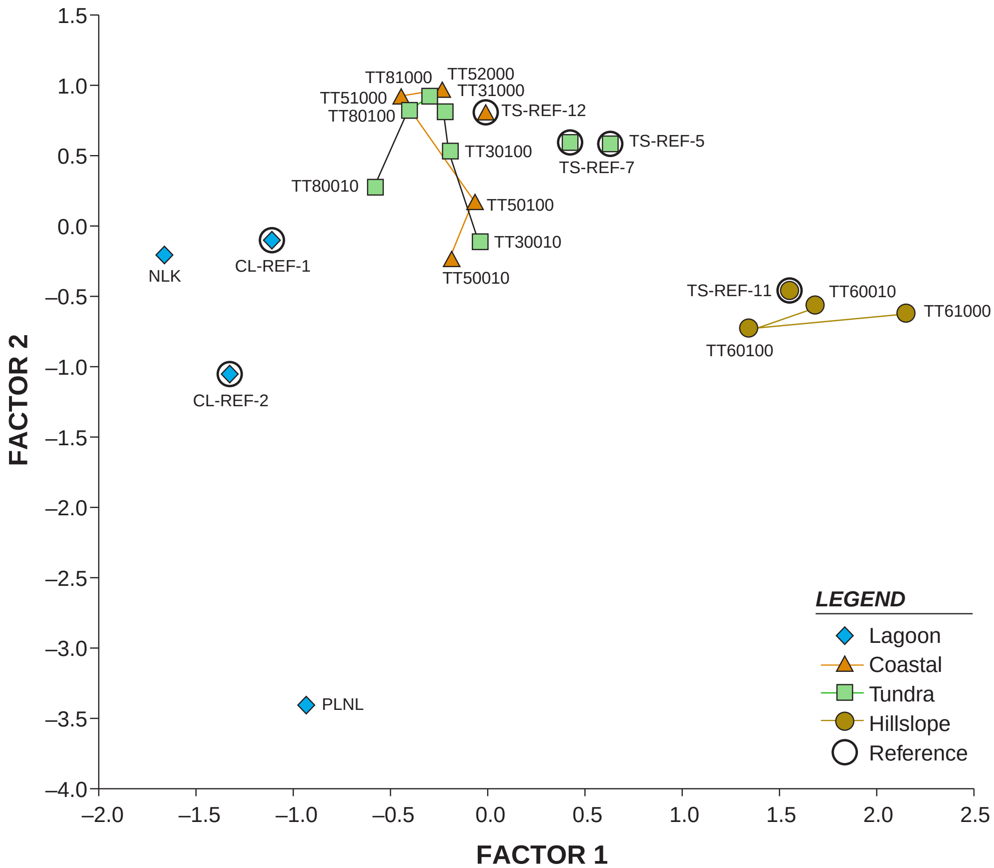

Coefficients for standardized factor scores after rotation

	Factor 1	Factor 2
Richness	**0.279**	–0.140
Deciduous shrubs	**0.236**	–0.016
Lichen	**0.213**	–0.031
Moss	**0.178**	–0.006
Diversity	0.137	0.134
Forbs	0.044	**–0.252**
Evergreen shrubs	–0.016	**0.200**
Unvegetated	–0.026	**–0.216**
Evenness	–0.063	**0.272**
Litter	–0.170	**0.247**
Graminoids	**–0.225**	0.062
Eigenvalue	3.622	3.490
Variance	32.9%	31.7%
Total variance	32.9%	64.7%
Before rotation:		
Eigenvalue	4.816	2.297
Variance	43.8%	20.9%
Total variance	43.8%	64.7%

Figure 6-5. Factors 1 and 2 from principal component analysis of high-level vegetation community variables with Varimax rotation

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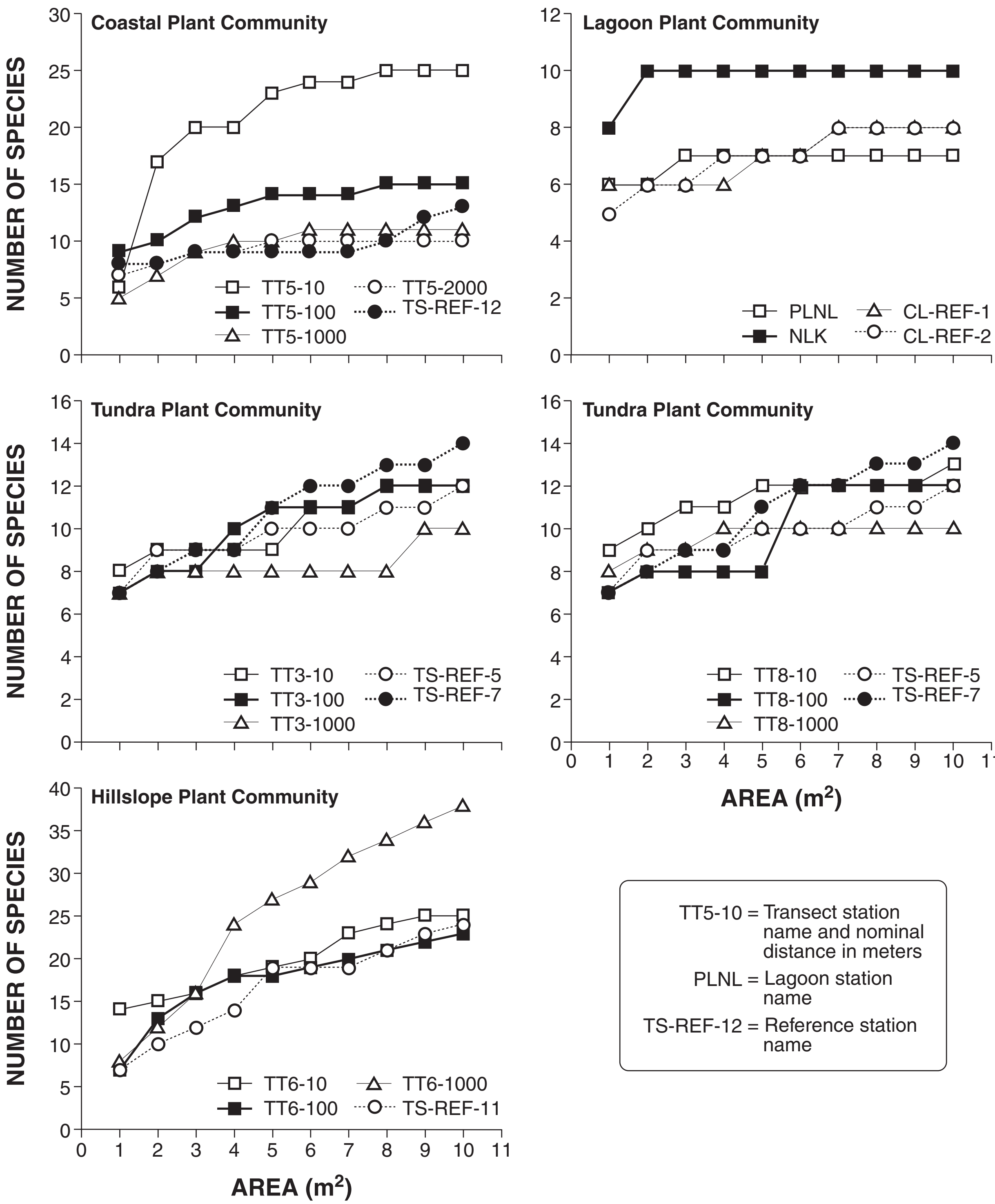
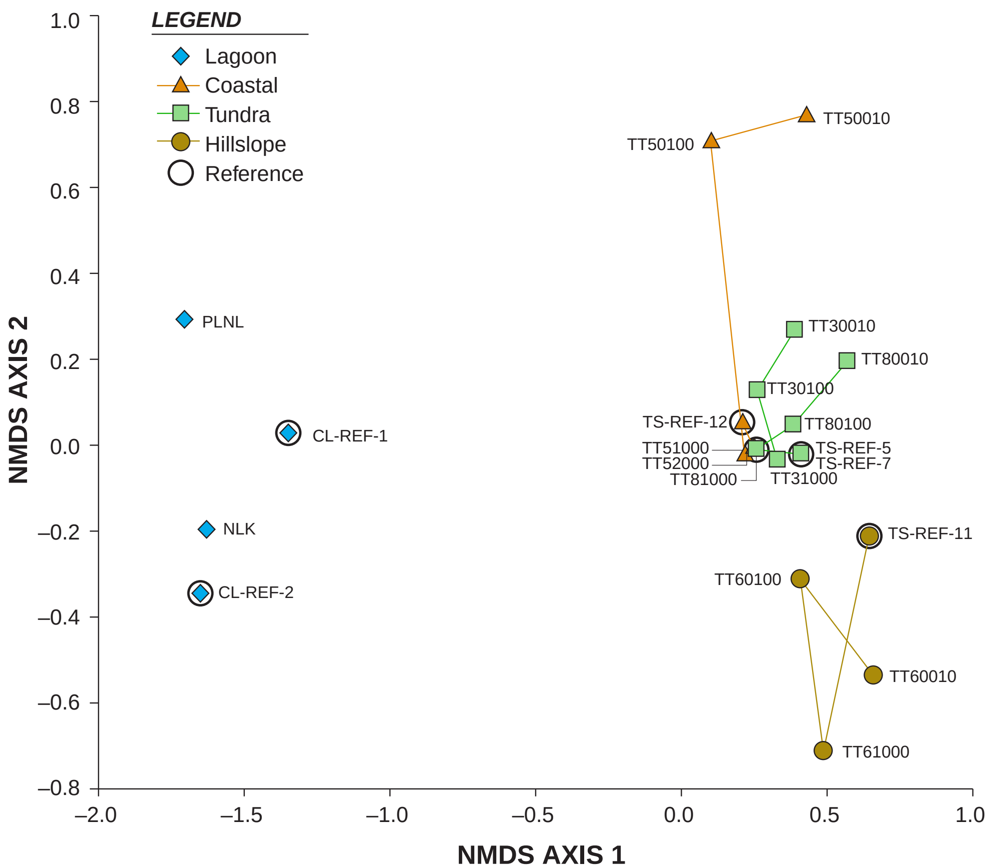


Figure ME1. Species area curves for plant community surveys



Coefficients of NMDS axes

		Axis 1	Axis 2
Deciduous Shrubs	*Salix planifolia pulchra*	0.725	0.360
	Salix lanata richardsonii	0.646	–1.452
	Salix polaris	0.562	1.870
	Vaccinium uliginosum alpinum	0.556	–0.209
	Betula nana exilis	0.535	–0.052
	Rubus chamaemorus	0.440	0.252
	Salix ovalifolia	0.336	0.321
Evergreen Shrubs	*Arctostaphylos alpina*	0.878	–1.284
	Empitrum nigrum hermaphorditum	0.552	–0.136
	Ledum palustre decumbens	0.456	0.020
	Vaccinium vitis-idaea minus	0.418	–0.064
	Andromeda polifolia	0.418	0.309
Forbs	*Pyrola grandiflora*	0.800	–1.426
	Equisetum arvense	0.730	–1.161
	Arnica lessingii lessingii	0.684	0.080
	Saussurea angustifolia	0.670	–1.663
	Pedicularis labradorica	0.658	–0.731
	Polygonum bistorta plumosum	0.644	–1.710
	Saxifraga punctata	0.644	–1.710
	Petasites frigidus or hyperboreus	0.528	0.838
	Polemonium acutiflorum	0.195	1.751
	Valeriana capitata	0.174	1.418
	Stellaria laeta	0.121	1.727
	Stellaria crassifolia	–2.190	–0.267
	Hippuris vulgaris	–2.278	0.664

		Axis 1	Axis 2
Graminoids	*Carex microchaeta*	0.875	–1.153
	Festuca altaica	0.827	–1.212
	Poa sp.	0.765	–1.104
	Arctagrostis latifolia var. latifolia	0.757	–0.612
	Caryx bigelowii	0.596	–0.200
	Eriophorum vaginatum	0.406	0.236
	Poa lanata	0.379	1.810
	Arctagrostis latifolia var. arundinaceae	0.268	1.774
	Eriophorum angustifolium subarcticum	-0.869	0.549
	Arctophila fulva	–1.919	0.108
	Carex aquatilis	–1.931	–0.524
	Deschampsia caespitosa	–2.156	–0.327
	Dupontia fischeri psilosantha	–2.156	–0.295
	Calamagrostis deschampsioides	–2.236	–0.663
	Lichen	0.504	–0.424
	Moss	0.136	–0.217
Litter	Broadleaf litter	0.364	0.044
	Dry blades	–0.023	0.214
	Detritus/fines	–2.142	–0.283
	Littoral matter	–2.224	–0.495
Unvegetated	Road gravel	0.629	1.020
	Bare ground	0.562	0.859
	Rock	–0.994	–0.584
	Water	–1.498	0.583
	Sand/gravel	–2.240	–0.716

Stress = 6.736

Figure ME2. Axes 1 and 2 from nonmetric multidimensional scaling analysis of vegetation species percent cover data

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