Kiggavik Project
Final Environmental Impact Statement

Tier 3 Technical Appendix 8A: Ecological and Human Health Risk Assessment

September 2014

History of Revisions

Revision Number	Date	Details of Revisions
01	December 2011	Initial release Draft Environmental Impact Statement (DEIS)
02	September 2014	FINAL Environmental Impact Statement

Table of Contents

List of Tables

List of Figures

Attachments

1 Introduction

1.1 Background

The Kiggavik Project includes a series of uranium deposits located in the Kivalliq region of Nunavut, approximately 80 km west of the community of Baker Lake. The Kiggavik Project includes two properties, Kiggavik and Sissons, which are associated with resources representing approximately 51,000 t U with a grade of approximately 0.46%. It is recognized that with continued exploration in the vicinity of the known economic reserves, there is the potential for additional ore deposits to be identified over the proposed life of the project. Ownership of the project is through a joint venture comprising AREVA Resources Canada Inc (ARC), UG Canada (collectively, "AREVA"), Japan-Canada Uranium Co. Ltd. ("JCU") and Daewoo International Corp. ("Daewoo"). ARC is the operator of the project (AREVA 2007).

The following section presents a brief description of the Kiggavik Project and the relevant activities that may impact on environmental quality in the study area.

1.1.1 Project Activities

The Project encompasses the mining of five deposits: East Zone, Centre Zone and Main Zone at Kiggavik, and Andrew Lake and End Grid at Sissons. Mining will proceed via open pit methods at the three Kiggavik deposits. The East Zone pit is mined out prior to mill commissioning to provide an initial Tailings Management Facility. Following preparation of the East Zone TMF, the mill will be commissioned at Kiggavik and processing of stockpiled ore will commence. Mining continues at Centre Zone, followed by Main Zone. These pits will both be converted into TMFs once mining is complete. A purpose-built-pit has been included as a water storage reservoir, allowing significant volumes of site and stockpile drainage to be stored for use in the mill.

Mining of the two Sissons deposits incorporates underground mining via overhand drift-and-fill at End Grid and open pit mining at Andrew Lake.

On completion of mining and milling activities, the project site area will be returned as close as practical to its natural state. The decommissioning plans include demolition and removal of site facilities and cleaning of all site areas that may have become contaminated. Closure of the three TMFs will include the placement of a layer of waste rock to promote consolidation followed by a layer of sand and an erosion barrier of waste rock over the tailings. The surface of the final cover will be graded to blend into the existing topography. Closure of the waste rock pile will include the

placement of a layer of compacted waste rock. The final surface of the waste rock pile will be regraded to blend into the existing topography and to allow for wildlife access.

For the Kiggavik Project, reagents, fuel and supplies will be containerized and shipped to Baker Lake via barge and then transported to site by truck over a winter road. The uranium product (yellowcake) will be shipped out to southern Canada by airlift. Power will be generated onsite using diesel generators.

1.2 OBJECTIVES

The purpose of the assessment is to determine the level of effect on the local environment (including people) for constituents of potential concern for the proposed Kiggavik Project.

The study included four major components as follows:

- Atmospheric dispersion modelling;
- Aquatic environment effects assessment;
- Ecological risk assessment; and
- Human health risk assessment.

The atmospheric dispersion modeling assessment was undertaken to evaluate the potential effects of mining, milling and associated activities on standard pollutants (SO2 and NO2), radon-222 levels, total suspended particulate (TSP) and associated levels of metals and radioactive constituents in the atmospheric environment. The results of this work are presented in a separate report. Predicted atmospheric levels were exported from the air dispersion model for input to the human health and ecological risk assessment.

The aquatic environment effects assessment was undertaken to predict downstream water and sediment quality for the effluent release scenarios selected for the assessment.

Figure 1.2-1 identifies which of the Tier 2 reports the information is either obtained (air concentrations are obtained from Volume 2) or where information from this report is reported (water and sediment concentrations in Volume 5; soil, vegetation and wildlife are in Volume 6; human health in Volume 8).

The ecological risk assessment was conducted to assess potential effects of the project on the atmospheric and aquatic environments and key ecological species (e.g., aquatic plants, fish, caribou). It was not intended for the assessment to consider all potentially affected ecological species, but rather to focus on selected valued ecosystem components (VECs).

The human health risk assessment was carried out to assess project effects from radon-222, TSP and standard pollutant levels at specific human receptor locations including the community of Baker Lake and at the on-site camp (Accomodation Complex). In addition, project effects on major dietary food resources were considered (e.g., on caribou and fish that may be obtained from the local study area).

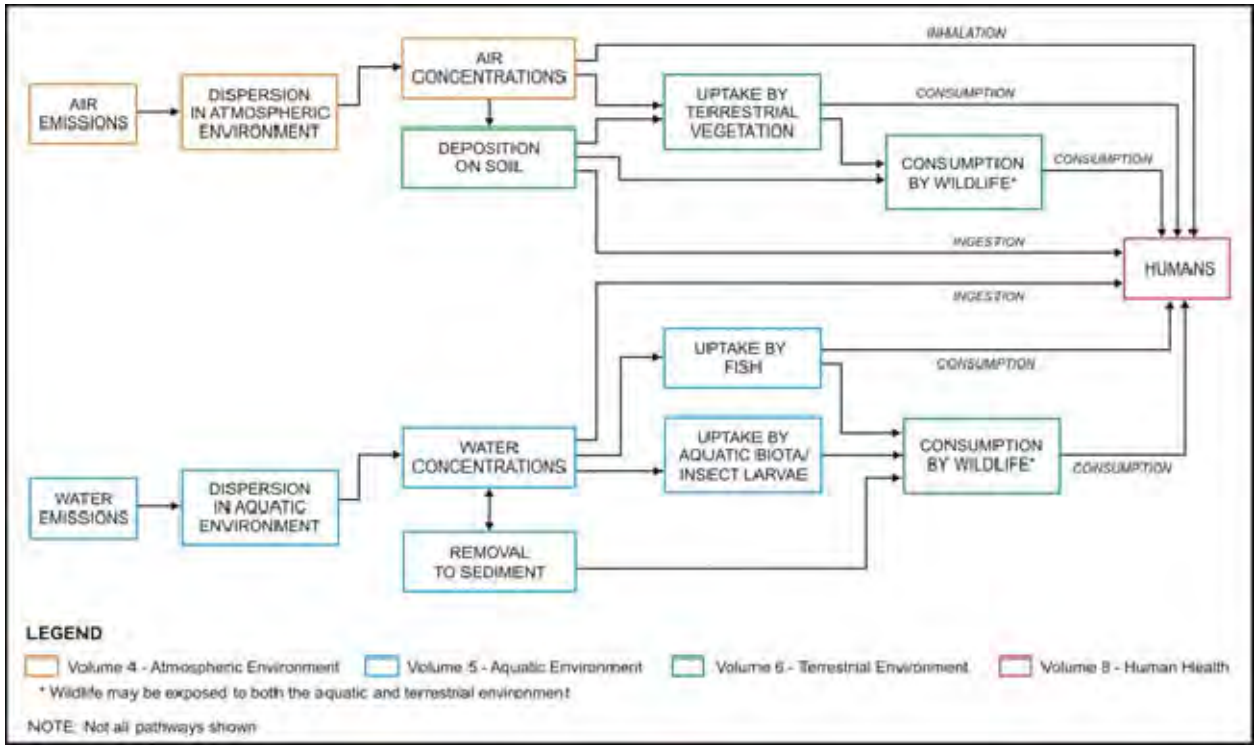


Figure 1.2-1 Linkages Between Environmental Components

1.3 OVERVIEW OF ASSESSMENT

The pathways framework considered in this assessment involves consideration of various pathways by which humans, animals and plants may be exposed to emissions from the Kiggavik Project. Thus, the pathways model or framework assesses the integrated contribution of emissions to the atmospheric and aquatic environments from all sources to determine environmental concentrations of constituents of potential concern and estimates exposure and dose levels for aquatic and terrestrial biota as well as humans.

Figure 1.3-1 presents a simplified schematic diagram of the pathways that are considered in the current study. As seen in the figure, emissions to both the atmospheric and aquatic environments have the potential to impact on humans, animals and plants.

To quantify potential exposures and effects into the future, mathematical modeling was conducted. To assess the potential effects resulting from Kiggavik Project effluent discharges on the aquatic environment, source constituent concentrations and flows, receiving environment flow monitoring results and baseline water and sediment quality data were combined to predict changes in water and sediment quality in the watershed downstream from the Kiggavik Project discharge locations. Changes in constituent concentrations in the receiving environment are a result of two very general processes: mixing of effluent with freshwater flow in the watershed; and, removal of constituents to sediments on settling solids and by diffusive exchange. Diffusive exchange is a two-way process, that is, constituents may diffuse from the water column to the sediment porewater, or from the sediment porewater to the water column depending on the concentration gradient.

To assess the fate of constituents in the surface waters and sediments the LAKEVIEW model was used. The LAKEVIEW model was developed and applied by SENES Consultants Limited on several other similar assessments. The model was modified for this application to take into account the effects of extensive ice formation in the area lakes on lake water quality and to take into account the fact there is no flow in the study area for six to eight months of the year. The model is developed within a probabilistic framework, which permits uncertainty in the magnitude of many of the model input parameters to be explicitly taken into account. The model was run probabilistically for 200 trials.

For this application, the model was run using monthly time steps to assess the effects of variations in source discharge loads and watershed flows on receiving water quality. At the Kiggavik Project, the treated effluent discharge is expected to comprise treated effluent from a conventional water treatment plant (WTP) and permeate water from a reverse osmosis (RO) system. Only treated effluent in excess of process needs will be discharged. The monthly time steps were deemed to be appropriate for simulation of receiving water quality since day-to-day variations in flow and effluent quality are evened-out in the water treatment plant.

Air concentrations were determined using the CALPUFF/CALMET model system. Estimates of the air concentrations from emissions to air from the Kiggavik Project were modelled separately, and output results from the air dispersion modeling were then entered into the pathways model. The CALPUFF/CALMET model system is recognized by regulatory agencies and is well suited to this type of application.

The pathways modeling was undertaken using the INTAKE model. The INTAKE model was developed by SENES Consultants Limited based on work performed by the firm in development of the Uranium Tailings Assessment Program (UTAP) pathways model in the 1980's (SENES 1984; 1985; 1986; 1987). The applicability of this model for simulating the exposures and impacts of operational discharges from uranium mines has been demonstrated on several projects in northern Saskatchewan. This model was then used to estimate exposure to both human and ecological receptors that could potentially be affected by the Kiggavik Project. The predicted exposures were

then combined with toxicological benchmarks to evaluate the potential adverse effects on humans and ecological receptors. The pathways model was also completed within a probabilistic framework.

As seen from the above discussion there are essentially two types of information needed to evaluate the potential adverse effects on humans or ecological receptors. The first type of information consists of inputs that are necessary for the pathways models and the second type of information needed for the assessment is effects-based estimates, which are used to gauge potential effects, or risk to identified receptors.

Site-specific data was used as much as possible to derive model inputs. In this regard, site-specific and local data were used to derive model inputs (e.g., meteorological data from Baker Lake, local surface hydrology measurements, limnological data on the area lakes, local surface water and sediment quality data, information on wildlife use of the study area, environmental samples collected from the area). Specifically, the following data were considered:

- Meteorological data;
- Air quality data;
- Surface hydrology and limnological data;
- Surface water and sediment quality data;
- Sediment extraction data;
- Aquatic vegetation data;
- Fish data;
- Soil quality data;
- Terrestrial vegetation data;
- Bird and animal data;
- Ecological survey data; and
- Human dietary survey data.

Where site-specific information was not available, information from other similar operations was used. For example, site-specific information was not available for estimating constituent transfer to environmental media (e.g., transfer from air or water to vegetation or transfer to food sources to herbivores, omnivores or carnivores). In these instances, data from other regions (e.g., from work carried out northern Saskatchewan) were used. In other cases, data reported in the scientific literature were used as is customarily done in ecological and human health risk assessments.

Measurement endpoints were chosen to represent key ecological receptor population and community characteristics of growth, reproduction and abundance. Measurement endpoints are commonly selected at the individual level of biological organization, and are typically based on exposure responses that represent key population and community characteristics, such as reproduction and abundance. Such measurement endpoints are commonly based on literature-derived toxicity dose-response relationships, examined through laboratory experimentation (i.e., the

response of a particular organism to a certain level of exposure). When derived from toxicity studies, such measurement endpoints are often referred to as toxicity benchmarks.

For humans, appropriate toxicological benchmarks were selected from Health Canada, Canadian Nuclear Safety Commission and the U.S. EPA.

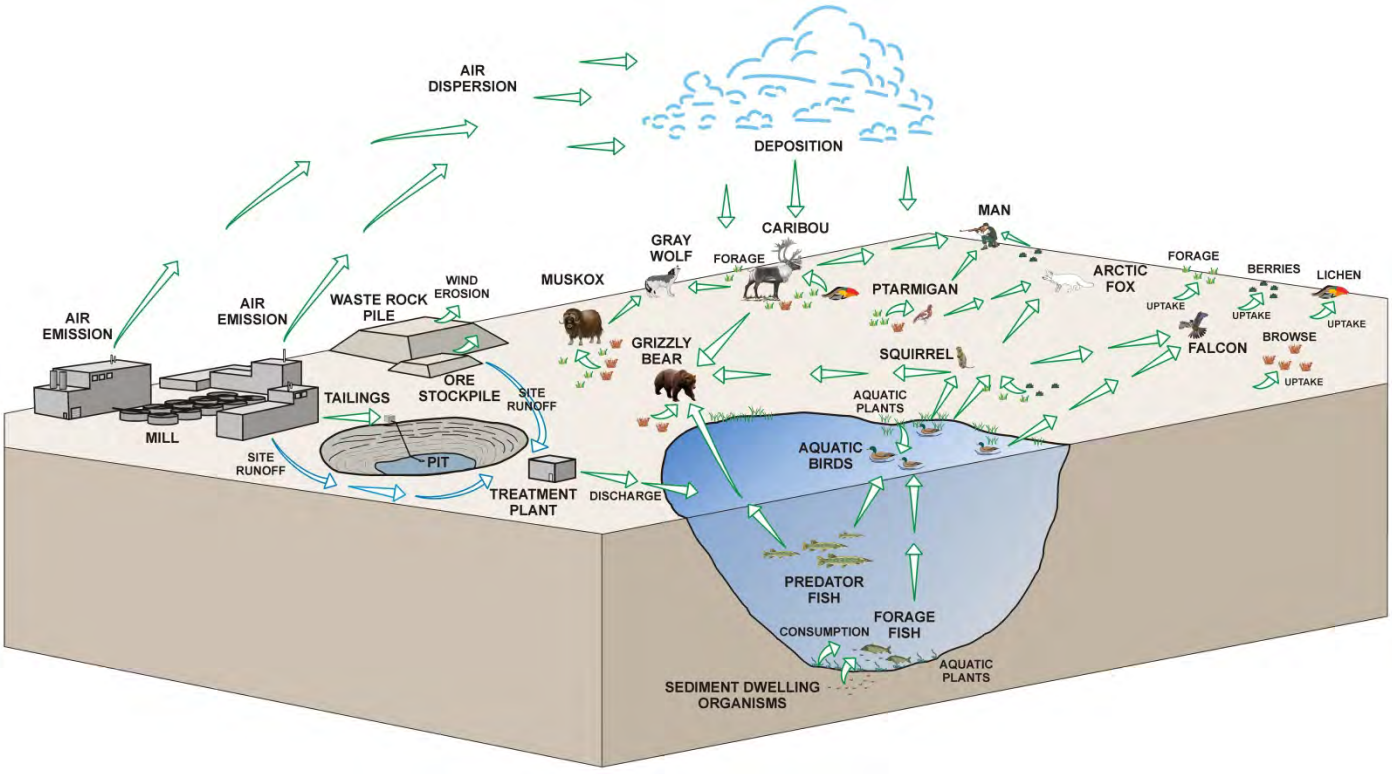


Figure 1.3-1 Simplified Diagram of Pathways Considered in the Environmental Assessment

1.4 REPORT STRUCTURE

Section 2 – Conceptual Model describes the main features of the assessment, including the selection of valued ecosystem components (VECs), selection of constituents of potential concern (COPC), and a description of the various models used in the assessment.

Section 3 – Site Characterization provides the baseline site data used for the assessment and describes the atmospheric and liquid emissions.

Section 4 - Receptor Characterization identifies the aquatic and terrestrial species selected for inclusion in the risk assessment, as well as the human receptors (i.e., adults and children) in the study area.

Section 5 – Exposure Assessment describes the pathways model used to predict the fate of constituents (metals and radionuclides) in the environment, including their uptake by aquatic and terrestrial species, removal to lake sediments. The bases for the transfer factors used in the pathways model are discussed.

Section 6 - Toxicity Assessment details the toxicity benchmarks used in the assessment to characterize the risks of potential effects on the health of ecological species and humans.

Section 7 – Receiving Environment Quality discusses the results of the water and sediment quality modelling that was conducted. Air, water, sediment, soil and vegetation quality predictions are presented and compared to available guidelines where applicable.

Section 8 - Ecological Risk Assessment presents the results of the pathways modelling and risk assessment for ecological receptors.

Section 9 - Human Health Risk Assessment presents the results of the pathways modelling and risk assessment for human receptors.

Section 10 - Summary and Conclusion - Implications of the Assessment are discussed in the context of observations of the current state of the ecosystem in the area of the Kiggavik Project. The significance of adverse effects on human health or the environment is discussed.

2 Conceptual Model

This assessment was undertaken within a pathways framework which involves consideration of humans, animals and plants which may be exposed to emissions from the Kiggavik Project. Thus, the assessment integrates the contribution of emissions to the atmospheric and aquatic environments from all sources to determine the potential for an adverse environmental effect.

2.1 Site Description

The Kiggavik and Sissons deposits are located within the Anigaq River watershed which comprises a portion of the Hudson Bay drainage basin. The Anigaq River drains into the western edge of Baker Lake, which in turn drains into Chesterfield Inlet and finally into Hudson Bay. Adjacent watersheds include Thelon River and Qinguq Creek, both of which drain into Baker Lake.

There are three hydrometric data stations within the region: Qinguq Creek near Baker Lake, Anigaq River below Audra Lake and Thelon River at the outlet of Schultz Lake. Table 2.1-1 presents the stations and watersheds details and long-term average data (Appendix 5A).

At the Anigaq River and Qinguq stations, there is no flow throughout the winter months. Peak flows are attributed to spring melts, with smaller peaks occurring in late summer after rainfall events. At the Qingaq station, seventy percent of outflow volume occurs in June and July. Flow in the Thelon River occurs year round in contrast to the much smaller Anigaq and Quinguq watersheds.

The Anigaq River watershed has been further sub-divided into sub-watersheds that correspond to locations of lake outflow. These sub-watersheds include the Boulder Lake, Lower Lake, Caribou Lake and Willow Lake systems draining into the Judge Sissons Lake and the Skinny Lake and Kavisilik Lake systems draining into Audra Lake. Gerhard Lake and Squiggy Lake, located outside of the Anigaq River system, drain into Aberdeen and Schultz Lakes. These lakes are drained by the Thelon River, which discharges into Baker Lake.

The sub-watershed falling within the study area in the current assessment is the system relevant to the water management system proposed for the Kiggavik Project, which is Judge Sissons Lake (see Figure 2.1-1).

The monthly distribution of flows was estimated based on available monthly discharge records/estimates for Qinguq Creek (see Table 2.1-2). The annual mean stream discharge estimates for Judge Sissons Lake shown in Table 2.1-3.

Table 2.1-1 Summary Data from Hydrometric Stations Located within the Study Area

Station ID	Station Name	Zone	Location X/Y	Drainage Area (km^2)	Period of Record	Mean Annual Flow (m^3/s)	Mean Annual Runoff ($m^3/s/km^2$)	Mean Flow During Open Water Period (m^3/s)	Mean Runoff During Open Water Period ($m^3/s/km^2$)
06MA002	Quinguq Creek near Bake Lake	14 W	629926/ 7128166	432	1969-1994	2.73	0.0063	6.00	0.0139
06MA007	Anigaq River below Audra Lake	14 W	617197/ 7123147	2740	1984-1994	17.0	0.0062	36.36	0.0133
06MA006	Thelon River below outlet of Schultz Lake	14 W	594017/ 7186013	152000	1983-2005	926	0.0061	926	0.0061
Reference: Appendix 5A									

The total and under-ice lake volumes are also shown in Table 2.1-3. These volumes have been estimated based on existing bathymetric data or mean depth and surface area data. The under-ice volume of lakes have been calculated for lakes deeper than two meters (the mean maximum thickness of winter ice depth for lakes in the vicinity of the Kiggavik Project is estimated to be 225 cm).

Table 2.1-2 Mean Monthly Flow Distribution in Qinguq Creek (1969-1994)

Month	Mean Monthly Discharge (m^3/s)	Total Volume (m^3)	Percentage of Annual Volume
Jan	0	0	0.0%
Feb	0	0	0.0%
Mar	0	0	0.0%
Apr	0	0	0.0%
May	0.7	1,842,739	2.4%
Jun	16.9	43,804,800	56.1%
Jul	5.6	15,025,824	19.2%
Aug	2.5	6,669,216	8.5%
Sep	3.6	9,227,520	11.8%
Oct	6	1,529,366	2.0%
Nov	0	28,512	0.0%
Dec	0	0	0.0%
Total	-	78,127,977	100.0%
Annual	2.6	82,939,680	-
Reference: Appendix 5A			

Table 2.1-3 Characteristics of Lakes Relevant for this Study

Lake	Drainage Area to Lake Outlet (km2)	Mean Annual Discharge (m3/s)	Surface Area (ha)	Mean Depth (m)	Maximum Depth (m)	Total Volume (m3)	Volume of Water Below 2m Contour (m3)	Retention Time (yr)
Judge Sissons Lake	704.6	4.289	9550	4.6	17	439,000,000	264,000,000	3.25
Notes: - no data								
Reference: Appendix 5A								

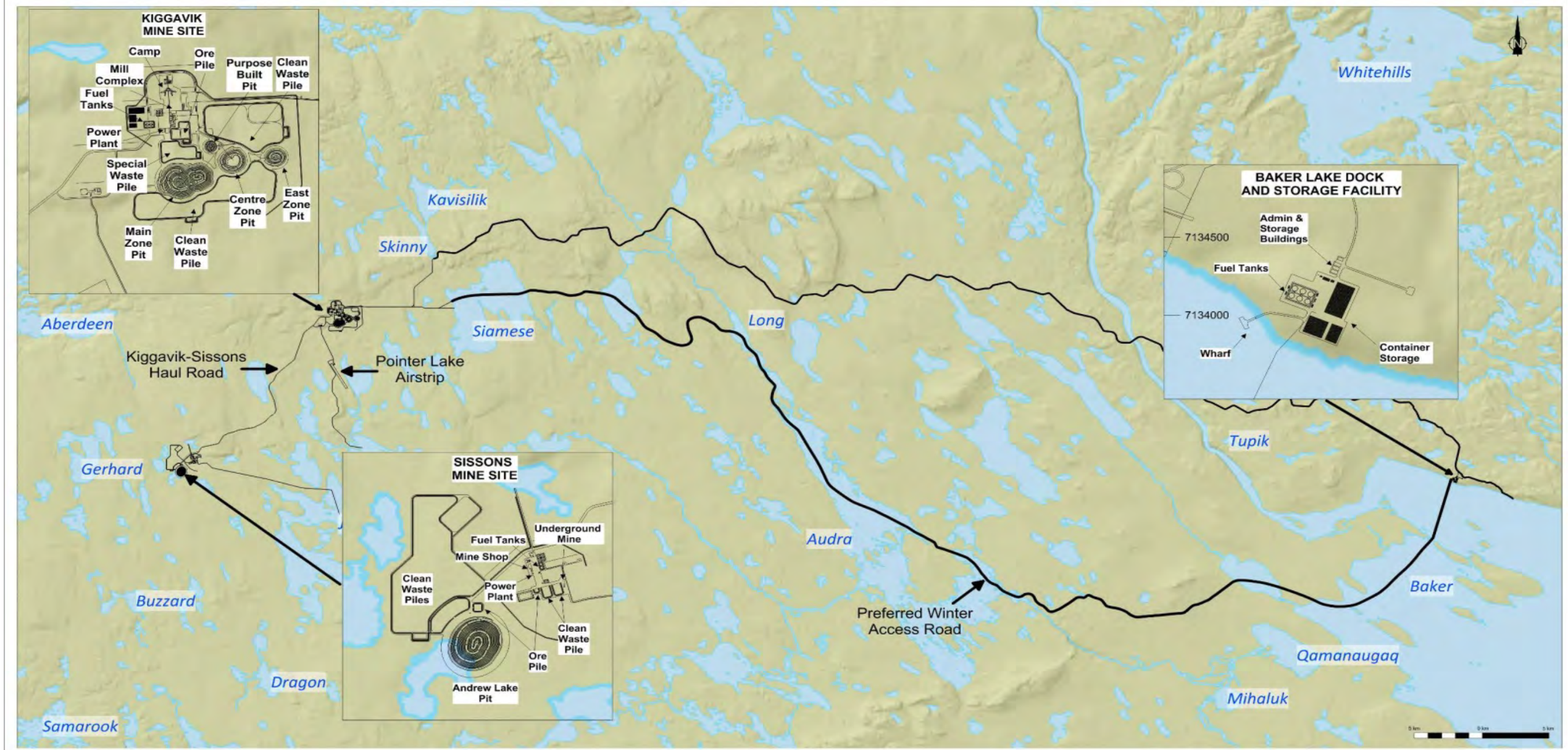


Figure 2.1-1 General Layout of the Kiggavik Project Site

2.2 Selection of Valued Ecosystem Components

Valued Ecosystem Components (VECs) are those aspects of the environment considered to be of vital importance to a particular region or community, including:

1. Resources that are either legally, politically, publically, or professionally recognized as important, such as parks, land selections, and historical sites;
2. Resources that have ecological importance; and,
3. Resources that have social importance (NIRB, 2007).

Valued Socio-Economic Components (VSECs) are those aspects of the socio-economic environment considered to be of vital importance to a particular region or community, including components relating to the local economy, health, demographics, traditional way of life, cultural well-being, social life, archaeological resources, existing services and infrastructure, and community and local government organizations (NIRB, 2007).

All environmental components were assessed with respect to specific features of the natural environment (e.g., water quality or air quality) and their roles in providing pathways and mechanisms for effects on the VECs based on the inter-relationships of the environmental components.

Generally, VEC selection considers the following:

- Abundance in the Site, Local and Regional Assessment Areas;
- Ecological importance - position in the food web and relative contribution to productivity;
- Baseline data availability - sufficient information should be available to allow a reasonable evaluation of effects;
- Native species, without setting aside the possibility to consider exotic species;
- Exposure - the VEC should have some degree of exposure to the “stressors” produced by the Project Works and Activities;
- Sensitivity - the VEC should be sensitive to the “stressors” produced by the Project Works and Activities;
- Ecological health – potential to affect the growth or sustainability of biota;
- Human health – potential to affect human health as represented by air quality guidelines, water quality criteria and guidelines and radiation dose to the public;
- Socio-economic importance – e.g., value as commercial, recreational or subsistence; inherent aesthetic value, including being of educational or scientific interest;
- Conservation status of Species at Risk - specifically protected by law, designated as rare, threatened, endangered, or of special concern;
- Traditional and current importance to people; and,
- Cultural and heritage importance to society.

The relevant VECs and VSECs selected for this component of the assessment are:

- Terrestrial environment, including soils;
- Surface water quality;
- Sediment quality;
- Freshwater aquatic environment - aquatic biota (including fish, aquatic macrophytes, benthic invertebrates, and other aquatic organisms);
- Vegetation;
- Terrestrial wildlife, including representative terrestrial mammals (i.e., caribou, muskoxen, wolverine, grizzly bears, wolves and less conspicuous species that may be maximally exposed to contaminants);
- Birds including raptors, migratory birds and seabirds, and their habitat; and
- Health of public.

An important consideration in an ecological risk assessment is the determination of the presence or absence of species at risk. The presence of species at risk can influence the choice of indicator species and its associated toxicity benchmark. Species at risk present at or near the Kiggavik Project area were identified (Appendix 6C) considering species listed by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC), Species at Risk Act (SARA), and the Canadian Endangered Species Conservation Council (CESCC).

Table 2.2-1 presents the species observed in the Kiggavik area that are at risk or of concern according to the relevant territorial and federal listings. A number of bird and mammal species were identified as regionally 'Sensitive' in Nunavut by the CESCC (2006). Although providing no legal protection, CESCC's status ranks are useful in identifying species of higher priority of study for the territory. For the proponent, this means more careful and particular consideration of the potential effects of the project on the species identified. Two mammals (grizzly bear and wolverine) and two birds (peregrine falcon and short-eared owl) have been identified by the COSEWIC as a special concern and are included on Schedule 3 under the federal SARA. However, none of these species are listed under Schedule 1 of the SARA. If a Schedule 1 species was identified, special measures to protect and recover such a listed species would be required.

Table 2.2-2 provides a summary of the selection of VECs/ indicator species used in the wildlife component of the ecological risk assessment and the rationale for the rationale for its selection.

Table 2.2-1 Sensitive Species or Species at Risk Observed in the Kiggavik Project Area

Wildlife Species	Scientific Name	Nunavut Status Rank[(a)]	COSEWIC Schedule 2 & 3 Status[(b)]	SARA Schedule 1 Status[(c)]
Mammals				
Grizzly Bear	*Ursus arctos*	Sensitive	Special Concern	No status
Wolverine	*Gulo gulo*	Secure	Special Concern	No status
Birds				
American golden plover	*Pluvialis dominica*	Sensitive	No status	No status
American tree sparrow	*Spizella arborea*	Sensitive	No status	No status
Golden eagle	*Aquila chrysaetos*	Sensitive	No status	No status
King eider	*Somateria spectabilis*	Sensitive	No status	No status
Peregrine falcon	*Falco peregrines* ssp. *tundrius*	Secure	Special Concern	No status
Ruddy turnstone	*Arenaria interpres*	Sensitive	No status	No status
Semi-palmated sandpiper	*Calidris pusilla*	Sensitive	No status	No status
Short-eared owl	*Asio flammeus*	Sensitive	Special Concern	No status
Snow bunting	*Plectrophenax nivalis*	Sensitive	No status	No status
White-crowned sparrow	*Zonotrichia leucophrys*	Sensitive	No status	No status
Wilson's snipe	*Gallinago delicata*	Sensitive	No status	No status

Notes:
a Canada wild species status ranks (CESCC 2006)
b COSEWIC 2009 Internet site
c SARA 2009 Internet site

Reference: Appendix 6C

Table 2.2-2 Terrestrial Ecological Receptors Included in the Assessment

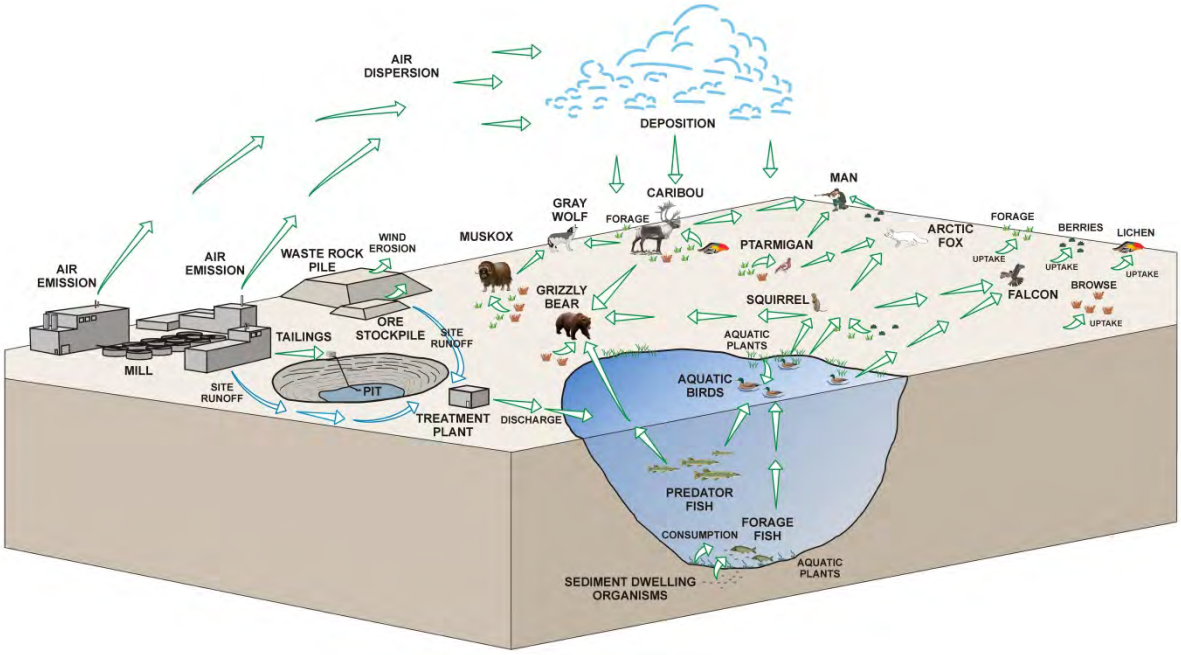
Receptor	Rationale	Similar Species
Herbivore Mammals		
Barren-ground Caribou	Important species for human diet and northern ecosystem; project area is within post-calving grounds	Other large herbivore mammals with a lichen diet
Brown Lemming	Common species; food source for fox, wolf, falcon, etc. Lemmings eat forage	Other small herbivore mammals, such as voles
Muskox	Important species for area, human diet	Other large herbivore mammals
Arctic ground Squirrel	Common species; could be human food source	Other small herbivore mammals, such as arctic hare
Omnivore Mammals		
Grizzly Bear	Important species for area: special concern and sensitive species	Other omnivore mammals

Table 2.2-2 Terrestrial Ecological Receptors Included in the Assessment

Receptor	Rationale	Similar Species
Insectivore Mammals		
Masked Shrew	Common species in riparian/wetland areas; insectivore	Other small insectivores
Carnivore Mammals		
Arctic Fox	Important species for trappers	Other scavenger mammals
Gray Wolf	Common species; dens in the project area	Other carnivore mammals
Wolverine	Important species, weasel family, but size of a small bear; has the largest home range of a carnivore of its size; carnivorous scavenger	Other small scavenger mammals
Aquatic Birds		
Long-tailed Duck	Diet includes benthic invertebrates (mollusks, crustaceans) and small fish	Other aquatic birds with primarily a benthic invertebrate diet
Red-breasted Merganser	Typical aquatic bird, ingestion of fish	Other aquatic birds with primarily a fish diet
Northern Pintail	Typical aquatic bird, ingestion of aquatic vegetation	Other aquatic birds with primarily aquatic vegetation diet
Raptor		
Peregrine Falcon	Species of special concern; carnivorous terrestrial bird	Other carnivorous terrestrial birds, such as short-eared owl
Terrestrial Birds		
Lapland Longspur	Common songbird of the Arctic Tundra; diet consists of insects and seeds.	Several other species of songbirds.
Rock Ptarmigan	No migration - resident year round ; could be human food source; herbivore terrestrial bird	Other terrestrial birds, such as grouse
Semipalmated Sandpiper	Represent shore-birds; breeding area within the Sissons Lease area; ingestion of benthic invertebrates	Other shore birds

2.3 Conceptual Site Model

The conceptual site model (Figure 2.1-1) for the Kiggavik Project graphically displays an overview of the relationship between the receptors, and of the physical-chemical processes that affect the fate and transport of COPC. It also illustrates some of the food web processes that occur.



Note: Conceptual only. Not all exposure pathways are indicated in the figure. See discussion in Section 4 for a complete list of exposure pathways.

Figure 2.3-1 Conceptual Site Model for the Kiggavik Project

2.4 Selection of Constituents of Potential Concern

SENES has extensive experience with uranium mining projects in northern Canada and the selection of constituents of potential concern (COPC) was guided by the expected quality of treated effluent based on this experience. Also, the results of treatability studies and the mineralogy of the ores were considered in the selection. The standard suite of COPC for these types of projects was selected: uranium and the uranium-238 decay series (thorium-230, lead-210, radium-226, and polonium-210), arsenic, cobalt, copper, lead, molybdenum, nickel, selenium, and zinc, as well as the standard water quality parameters ammonia, chloride, sulphate, and total dissolved solids (TDS). Cadmium was added to the standard list of COPC, as was hardness (as calcium carbonate). The general water quality parameters (ie., ammonia, chloride, sulphate, TDS, and hardness) were evaluated for water quality, because it is assumed that these parameters do not undergo any removal from the water

column to the sediments due to sorption. The other COPC (ie., uranium, uranium-238 decay series, arsenic, cadmium, cobalt, copper, lead, molybdenum, nickel, selenium, and zinc) were evaluated for water quality, sediment quality, and through pathways analysis to examine the potential exposure of ecological and human receptors.

Consitutents, such as aluminum, chromium, iron, mercury, and silver, were observed at elevated levels in the receiving environment; however, these constituents were not selected as COPC for the assessment because these constituents are not related to the project, thus it is not necessary to consider them in the assessment. The natural thorium decay chain (Th-228, etc.) was considered and found not to be an issue for this site. The levels of thorium measured in samples collected from the ore and waste rock from the site are within typical background levels and do not show any association with the ore. As the dose criteria are based on the incremental exposure from the project it is not necessary to include radionuclides that are not associated with the Kiggavik operations and thus the natural thorium series was not included.

2.5 Modelling Approach

The assessment of ecological and human receptor exposures for COPC involved estimation of constituent levels in environmental compartments (i.e., air, water, sediment and soil) and environmental media (e.g., aquatic and terrestrial vegetation and animals).

An overview of the approach taken to model constituent dispersion and movement through the aquatic, atmospheric and terrestrial environments, and for the estimation of intake or dose to the ecological and human receptors is provided in this section. Assumptions and equations for the various models are summarized in Section 2.6 (air dispersion model), Section 2.7 (aquatic dispersion model), Section 2.8 (aquatic biota model), Section 2.9 (terrestrial biota model), and Section 2.10 (human model). Details are provided in Appendix 4b (Air Quality),Attachment A (LAKEVIEW Model), and Attachment D(Ecological Receptor Characteristics).

Over the past three decades, SENES Consultants Limited (SENES) has developed extensive expertise in the areas of contaminant transport modelling and ecological and human health risk assessment. During the 1980s, SENES developed the Uranium Tailings Assessment Program (UTAP) on behalf of the Canadian government and the mining industry to simulate the geochemical behaviour of metals and radionuclides in uranium tailings and in the receiving environment (SENES 1985, 1986, 1987). The UTAP model was designed to simulate the movement of contaminants through the aquatic, atmospheric and terrestrial environments including uptake by plants and animals. The model was constructed within a probabilistic framework to permit uncertainty in many of the model parameters to be taken explicitly into account, and also included a feature to estimate exposure doses to people living within the airshed or watershed.

During the early 1990s, the original model was enhanced to include routines for the estimation of ecological risk and human health risk. At the same time, SENES participated in an international model validation study known as BIOMOVS, in which several models designed to quantify the transfer and bioaccumulation of metals and radionuclides were tested. The SENES model (INTAKE) was shown to compare well with the other models and has been applied on several mining projects to assess the environmental effects of both proposed new developments and decommissioning of existing facilities. In addition, SENES has applied pathways analysis and risk assessment tools on numerous industrial sites.

The INTAKE model was developed to permit calculations to be carried out in either a deterministic framework, using nominal values of the input parameters, or a probabilistic framework to explicitly take into account uncertainty in the input parameter values. For the Kiggavik Project, the calculations were conducted in a probabilistic framework. Table 2.5-1 lists parameter descriptions defining distributions used in the uncertainty analysis.

Table 2.5-1 Descriptors of Parameter Distributions Used in Uncertainty Analysis

Distribution	Distribution Descriptors
Constant (C)	Constant
Normal (N)	Mean Standard Deviation Minimum Maximum
Lognormal (LN)	Geometric Mean Geometric Standard Deviation Minimum Maximum
Uniform (U)	Minimum Maximum
Triangular (T)	Minimum Mode Maximum
Beta (B)	Minimum Maximum Alpha shape parameter Beta shape parameter

2.6 Air Dispersion Model

To evaluate the effects of the Kiggavik Project on the local and regional atmospheric environment, air dispersion modelling was performed using the CALMET/CALPUFF modelling package, a current, state-of-the-art dispersion model. The CALPUFF model was selected for this study because of the

model's ability to handle both complex meteorology and an array of multiple emissions sources from facilities and activities located over a large area. The meteorological conditions in the LAA and RAA are unique. As such, the CALMET/CALPUFF model set is better able to simulate the meteorology and dispersion in this area.

The air dispersion model has been described in detail in other documents (Appendix A Air Quality); however, a brief description of the air dispersion model as it relates to the pathways assessment is provided here. CALPUFF is a multi-layer, multi-species, non-steady-state puff dispersion model that can simulate the effects of varying meteorological conditions in time and space on COPC transport (Scire et al. 1999). CALMET is an advanced non-steady-state diagnostic meteorological model that produces hourly three-dimensional gridded wind fields from available meteorological, terrain and land use data (Scire et al. 2000). CALPUFF runs in conjunction with CALMET to estimate the concentration or deposition value for each source-receptor combination for each hour of input meteorology. It can calculate short-term averages such as 1 and 24-hour or annual averages for COPCs of interest. CALPUFF is also one of the air dispersion models currently accepted by the US EPA for regulatory compliance modelling. In this assessment, Version 6.326 of the CALMET model and Version 6.262 of the CALPUFF model were used (which are the most recent versions at the time the modelling was undertaken).

The CALMET meteorological model was used to simulate meteorological conditions in the study area (LAA and RAA). The CALMET model was run for a large modelling domain measuring 117 km in an east-west direction and 65 km in a north-south direction, with a grid spacing of 1 km.

One of the meteorological stations is located within the Local Assessment Area near the proposed Pointer Lake Airstrip, about 3 km south of the Accommodation Complex. It was installed in August 2009 and since then it has been recording surface temperature, relative humidity, wind speed and direction and atmospheric pressure data. Hourly surface wind speed and direction data from this station was used to develop the one year meteorological data set for air dispersion modelling. The meteorological dataset used in this assessment is for the period August 15, 2009 to August 14, 2010.

2.7 Aquatic Dispersion Model

The LAKEVIEW module of the INTAKE model was used to calculate water and sediment concentrations in Judge Sissons Lake (JSL). The LAKEVIEW dispersion model has been applied to several other uranium mining projects in northern Saskatchewan to simulate constituent transport and concentrations in the aquatic environment for input to a pathways model for evaluation of exposures to ecological species and humans. For this application, the LAKEVIEW model was modified to simulate the effects of extensive ice cover on Judge Sissons Lake (on the order of 2m thick) as well as the effects of prolonged periods (up to 8 months in any year) with no flow. A probabilistic framework was used for the 200 trials to simulate constituent transport and dispersion

through the segments of the lake. A description of the LAKEVIEW module is provided in Attachment A.

Important processes incorporated into the LAKEVIEW model include horizontal (lateral) and vertical transport of dissolved species, chemical and biochemical reactions in the sediment and in the water column, settling of particulate matter, and sediment exchange processes. LAKEVIEW incorporates a detailed computational protocol for estimating the flux of dissolved chemical species in and out of the sediment together with chemical reactions (reduction or oxidation) and solid phase and solid solution partitioning along with conventional sorption equilibrium. Robust and efficient methodologies for parameter estimation and parameter sensitivity analysis have also been incorporated into the program.

Where possible, site-specific data or data reported for similar environments (e.g., northern Saskatchewan) were used to characterize inputs to the LAKEVIEW model. These inputs include baseline water and sediment quality in the Kiggavik Project area and water-to-sediment distribution coefficients for estimation of constituent concentrations in sediment resulting from changes in concentrations in the water column of affected waterbodies.

Simulation results from the aquatic dispersion model were used in the pathways model to predict exposure to aquatic species and in the estimation of exposure to VECs and humans.

2.7.1 Characterization of Judge Sissons Lake

Judge Sissons Lake was divided into eight segments (Figure 2.7-1), which were defined as summarized in Table 2.7-1. The model takes into account the effects of ice formation on the concentrations of the COPC in both the water column and in sediments. Ice thickness of 2 m is typical of the study area and ice cover typically lasts 8 to 9 months per year. The effects of this natural occurrence are evident in the simulation results presented in this report, and, due to the difference in average depth of the Judge Sissons Lake segments, the COPC concentrations in each segment are affected differently by the ice cover. For example, JSL-1 is shallow compared to JSL-4 (average depth of 2.6 m in JSL-1 compared with 8.8 m in JSL-4) and therefore, there is very little free-flowing water in the winter months below the ice cover and the COPC concentrations in water increase more in JSL-1 relative to adjacent segments that are greater in depth.

Table 2.7-2 summarizes the parameter definition for the horizontal (lateral) transport of dissolved species between the water columns of the various Judge Sissons Lake segments. From Table 2.7-2, it can be seen that the connection between JSL-6 and JSL-7 and the connection between JSL-7 and JSL-8 are frozen solid during the winter months.

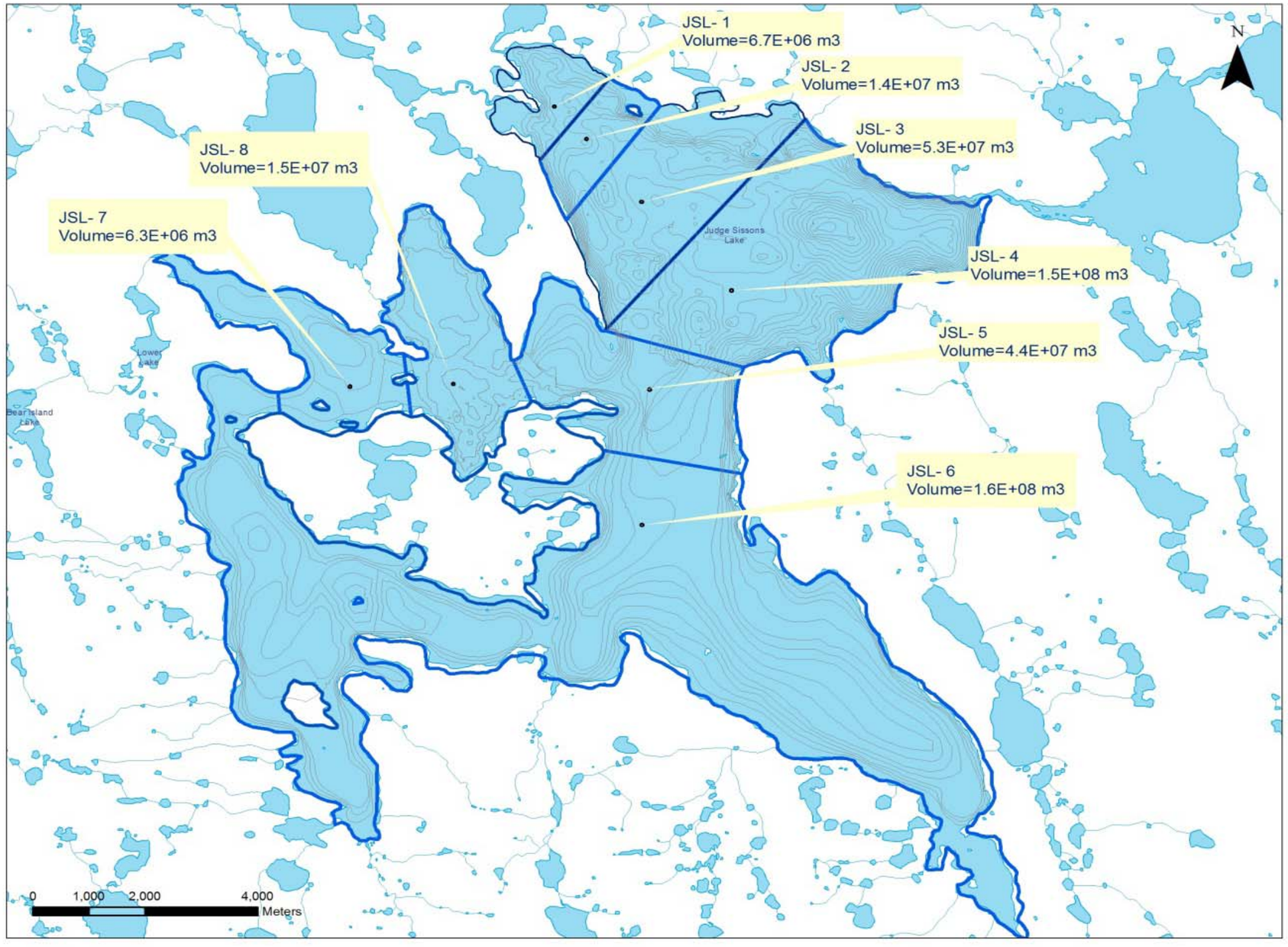


Figure 2.7-1 Judge Sissons Lake

Table 2.7-1 Summary of Judge Sissons Lake Segments

Description of Judge Sissons Lake Segments									
Parameter	**Units**	**JSL-1**	**JSL-2**	**JSL-3**	**JSL-4**	**JSL-5**	**JSL-6**	**JSL-7**	**JSL-8**
Volume	m^3	$6.7x10^6$	$1.4x10^7$	$5.3x10^7$	$1.5x10^8$	$4.4x10^7$	$1.6x10^8$	$6.3x10^6$	$1.5x10^7$
Area	m^2	$2.6x10^6$	$2.5x10^6$	$8.4x10^6$	$1.7x10^7$	$8.5x10^6$	$4.8x10^7$	$5.6x10^6$	$7.2x10^6$
Drainage Area (DA)	m^2	$1.9x10^8$	$5.8x10^6$	$1.5x10^7$	$3.1x10^7$	$2.3x10^7$	$2.7x10^8$	$8.4x10^7$	$8.4x10^7$
Cumulative DA	m^2	$1.9x10^8$	$1.9x10^8$	$2.1x10^8$	$7.0x10^8$	$4.6x10^8$	$2.7x10^8$	$8.4x10^7$	$1.7x10^8$
Average Depth*	m	2.6	5.6	6.3	8.8	5.2	3.3	1.1	2.1

Notes:
estimated from bathymetry of JSL completed in 2013
DA = drainage area
* - calculated as the volume/area

Table 2.7-2 Summary of Horizontal Dispersion

Judge Sissons Lake Segments – Horizontal Dispersion				
Segment	**Segment**	**Cross-Section Area Summer (m^2)**	**Cross-Section Area Winter (m^2)**	**Distance between Midpoints (m)**
JSL-1	JSL-2	9,000	5,200	850
JSL-2	JSL-3	18,000	13,000	1,600
JSL-3	JSL-4	39,000	29,000	2,300
JSL-4	JSL-5	22,000	18,000	2,700
JSL-5	JSL-6	16,000	11,000	2,700
JSL-6	JSL-7	260	0	2,100
JSL-7	JSL-8	1,000	0	1,900
JSL-5	JSL-8	1,700	378	3,500

Note: estimated from bathymetry of JSL completed in 2013.

2.7.2 Water and Sediment Distribution

It is assumed that the general parameters: chloride, sulphate and TDS do not undergo any removal from the water column to the sediments due to sorption, although there may be some exchange via diffusion. Likewise, while ammonia removal from the water column is modelled, either through conversion to nitrate or in the formation of new biomass, exchange with lake sediments is not explicitly modelled.

The LAKEVIEW model uses porewater-to-sediment distribution factor inputs to calculate water-to-sediment distribution factors, according to the equation (2.7-1):

$$Kd = Kd_{sed} \times FACT \quad (2.7\text{-}1)$$

where:

Kd = water-to-sediment distribution coefficient (m^3/g dw)
Kd_{sed} = porewater-to-sediment distribution coefficient (m^3/g dw)
FACT = relationship factor between Kd and Kd_{sed} (normal distribution with an average of 3, standard deviation of 1, minimum of 1, and maximum of 5).

The Kd is used to govern the distribution of COPC between the water and sediments. Kd values used in the assessment are provided in Section 5.1.1

2.8 Aquatic Biota Model

Modelling the uptake of radionuclides and metals by aquatic biota can be quite involved and requires consideration of several interactions. For example, direct uptake from water by aquatic vegetation, uptake from sediments by rooted vegetation, sedimentation and deposition of aquatic vegetation to lake sediment, decomposition and release of contaminants from lake sediments to the water phase, harvesting of aquatic vegetation by invertebrates, and predation of large fish on small fish may be considered in the modelling. The bioaccumulation factors used in the aquatic biota model are an aggregate representation of numerous pathways and mechanisms of radionuclide and metal transfer through the aquatic environment.

2.8.1 Fish

In the INTAKE model, the concentration in fish flesh is calculated as a function of the transfer factor and the concentration in the water phase from the relationship shown in equation (2.8-1):

$$C_{fish} = C_{water} \times TF_{fish} \quad (2.8\text{-}1)$$

where:

C_{fish} = concentration of constituent in fish reported as Becquerels per gram (wet weight) or grams per gram (wet weight) [Bq/g (wet weight) or g/g (wet weight)]
C_{water} = water concentration reported as Becquerels per cubic meter or grams per cubic meter [Bq/m^3 or g/m^3]
TF_{fish} = the water-to-fish transfer factor of radionuclide or metal (Bq/g (wet weight) per Bq/m^3) or (g/g (wet weight) per g/m^3)

The implicit assumption is that there is a linear relationship between the water and fish concentrations, which is not necessarily true. A review of water concentrations versus fish tissue concentrations for many metals indicate that metal concentrations in fish tissue tend to be independent of water concentrations at background levels and increase only after a certain concentration has been reached. This is consistent with observations by others (Brix et al. 2005). It has been postulated that for essential metals such as selenium, molybdenum and others, bioregulation or homeostasis is responsible for these observations (Toll Environmental 2005).

Therefore, a "hockey stick" model has been developed for the transfer of constituents in water to fish flesh, as shown in equation (2.8-2):

$$C_{fish} = C_{water} \times TF_{fish} \quad (2.8\text{-}2)$$

$$If\ C_{fish} < C_{fish-\min}, then\ C_{fish} = C_{fish-\min}$$

where:

C_{fish} = concentration of constituent in fish reported as Becquerels per gram (wet weight) or grams per gram (wet weight) [Bq/g (wet weight) or g/g (wet weight)]

$C_{fish\text{-}min}$ = minimum constituent concentration in fish as Becquerels per gram (wet weight) or grams per gram (wet weight) [Bq/g (wet weight) or g/g (wet weight)]

Transfer factors for fish are provided in Section 5.1.2.1. For the assessment of dose from radionuclides, concentrations in fish are predicted as described in equation 2.8-2 and this concentration is converted into an internal dose as shown in equation 2.8-3: $D-\text{int}_{fish} = C_{fish} \times DC_{\text{int}}$ (2.8-3)

where:

$D\text{-}int_{fish}$ = internal dose to fish reported as milliGray per day [mGy/d]

C_{fish} = concentration of constituent in fish reported as Becquerels per gram (wet weight) [Bq/g (wet weight)] {equation 2.8-2}

DC_{int} = internal dose coefficient reported as milliGray per day per Becquerels per gram (wet weight) [mGy/d per Bq/g (wet weight)]

Internal dose coefficients for ecological receptors are provided in Section 5.1.4.2

The assessment of dose from radionuclides for fish also considers the external dose. For predatory fish, the external dose is assumed to be entirely from radionuclides in water. For forage fish, the external dose is assumed to be the average of the dose from water and sediment, which is based on the forage fish immersed 50% in sediment and 50% in water (i,e, lying on the bottom of the lake). The external dose is calculated as shown in equation 2.8-4:

$$D-ext_{fish} = C_{water} \times DC_{ext} \div 1000000 \quad \text{\{predatory fish\}} \quad (2.8\text{-}4)$$

$$D-ext_{fish} = \left(C_{water} \times DC_{ext} \div 1000000\right) \times 0.5 + \left(C_{sed} \times DC_{ext}\right) \times 0.5 \quad \text{\{forage fish\}}$$

where:
$D\text{-}ext_{fish}$ = external dose to fish reported as milliGray per day [mGy/d]
C_{water} = water concentration reported as Becquerels per cubic meter or grams per cubic meter [Bq/m^3 or g/m^3]
1000000 = unit conversion factor for m^3 to g
C_{sed} = sediment concentration reported as Becquerels per grams [Bq/g]
DC_{ext} = external dose coefficient reported as milliGray per day per Becquerels per gram (wet weight) [mGy/d per Bq/g (wet weight)]

External dose coefficients for ecological receptors are provided in Section 5.1.4.2.

The total absorbed radiological dose for fish is calculated as the sum of the internal and external doses, as shown in equation 2.8-5:

$$D-tot_{fish} = D-ext_{fish} + D-\text{int}_{fish} \quad (2.8\text{-}5)$$

where:
$D\text{-}ext_{fish}$ = external dose to fish reported as milliGray per day [mGy/d]
$D\text{-int}_{fish}$ = internal dose to fish reported as milliGray per day [mGy/d]
$D\text{-}tot_{fish}$ = total absorbed dose to fish reported as milliGray per day [mGy/d]

For the evaluation of radiological dose to fish, the absorbed dose is multiplied by an appropriate radiation weighting factor to account for differences in the effects of alpha, beta and gamma radiation. This is achieved through the use of a relative biological effectiveness (RBE) factor (see Section 6.5.1), which is applied to the absorbed dose to calculate an equivalent dose, as shown in equation 2.8-6:

$$D-eq_{fish} = D-ext_{fish} + D-\text{int}_{fish} \times RBE \quad (2.8\text{-}6)$$

where:
$D\text{-}ext_{fish}$ = external dose to fish reported as milliGray per day [mGy/d]
$D\text{-int}_{fish}$ = internal dose to fish reported as milliGray per day [mGy/d]
RBE = relative biological effectiveness factor
$D\text{-}eq_{fish}$ = equivalent dose to fish reported as milliGray per day [mGy/d]

2.8.2 Aquatic Vegetation

In the INTAKE model, the concentration in aquatic plants is calculated as a function of the transfer factor and the concentration in the water phase from the relationship shown in equation (2.8-7):

$$C_{aqveg} = C_{water} \times TF_{aqveg} \qquad (2.8\text{-}7)$$

where:

C_{aqveg} = concentration of constituent in aquatic vegetation reported as Becquerels per gram (wet weight) or grams per gram (wet weight) [Bq/g (wet weight) or g/g (wet weight)]

C_{water} = water concentration reported as Becquerels per cubic meter or grams per cubic meter [Bq/m^3 or g/m^3]

TF_{aqveg} = the water-to-aquatic vegetation transfer factor of radionuclide or metal (Bq/g (wet weight) per Bq/m^3) or (g/g (wet weight) per g/m^3)

Transfer factors for aquatic vegetation are provided in Section 5.1.2.2. For the assessment of dose from radionuclides, concentrations in aquatic vegetation are predicted as described in equation 2.8-7 and this concentration is converted into an internal dose as shown in equation 2.8-8:

$$D-\text{int}_{aqveg} = C_{aqveg} \times DC_{\text{int}} \qquad (2.8\text{-}8)$$

where:

$D\text{-int}_{aqveg}$ = internal dose to aquatic vegetation reported as milliGray per day [mGy/d]

C_{aqveg} = concentration of constituent in aquatic vegetation reported as Becquerels per gram (wet weight) or grams per gram (wet weight) [Bq/g (wet weight) or g/g (wet weight)] {equation 2.8-7}

DC_{int} = internal dose coefficient reported as milliGray per day per Becquerels per gram (wet weight) [mGy/d per Bq/g (wet weight)]

Internal dose coefficients for ecological receptors are provided in Section 5.1.4.2.

The assessment of dose from radionuclides for aquatic vegetation also considers the external dose. For the leaves of aquatic vegetation, the external dose is assumed to be entirely from radionuclides in water. For the roots of aquatic vegetation, the external dose is assumed to be entirely from radionuclides in sediment. The external dose is calculated as shown in equation 2.8-9:

$$D-ext_{aqveg-leaf} = C_{water} \times DC_{ext} \div 1000000 \quad \{\text{leaves}\} \qquad (2.8\text{-}9)$$

$$D-ext_{aqveg-root} = C_{sed} \times DC_{ext} \quad \{\text{roots}\}$$

where:

$D\text{-}ext_{aqveg\text{-}leaf}$ = external dose to aquatic vegetation leaves reported as milliGray per day [mGy/d]

$D\text{-}ext_{aqveg\text{-}root}$ = external dose to aquatic vegetation roots reported as milliGray per day [mGy/d]

C_{water} = water concentration reported as Becquerels per cubic meter or grams per cubic meter [Bq/m^3 or g/m^3]

1000000 = unit conversion factor for m^3 to g

C_{sed} = sediment concentration reported as Becquerels per grams [Bq/g]

DC_{ext} = external dose coefficient reported as milliGray per day per Becquerels per gram (wet weight) [mGy/d per Bq/g (wet weight)]

External dose coefficients for ecological receptors are provided in Section 5.1.4.2.

The total absorbed radiological dose for aquatic vegetation is calculated as the sum of the internal and external doses, as shown in equation 2.8-10:

$$D-tot_{aqveg-leaf} = D-ext_{aqveg-leaf} + D-\text{int}_{aqveg} \quad \{\text{leaves}\} \quad (2.8\text{-}10)$$

$$D-tot_{aqveg-root} = D-ext_{aqveg-root} + D-\text{int}_{aqveg} \quad \{\text{roots}\}$$

where:

$D\text{-}ext_{aqveg\text{-}leaf}$ = external dose to aquatic vegetation leaves reported as milliGray per day [mGy/d]

$D\text{-}ext_{aqveg\text{-}root}$ = external dose to aquatic vegetation roots reported as milliGray per day [mGy/d]

$D\text{-}\text{int}_{aqveg}$ = internal dose to aquatic vegetation reported as milliGray per day [mGy/d]

$D\text{-}tot_{aqveg\text{-}leaf}$ = total absorbed dose to aquatic vegetation leaves reported as milliGray per day [mGy/d]

$D\text{-}tot_{aqveg\text{-}root}$ = total absorbed dose to aquatic vegetation roots reported as milliGray per day [mGy/d]

For the evaluation of radiological dose to aquatic vegetation, the absorbed dose is multiplied by an appropriate radiation weighting factor to account for differences in the effects of alpha, beta and gamma radiation. This is achieved through the use of a relative biological effectiveness (RBE) factor (see Section 6.5.1), which is applied to the absorbed dose to calculate an equivalent dose, as shown in Equation 2.8-11:

$$D-eq_{aqveg-leaf} = D-ext_{aqveg-leaf} + D-\text{int}_{aqveg} \times RBE \ \{\text{leaves}\} \quad (2.8\text{-}11)$$

$$D-eq_{aqveg-root} = D-ext_{aqveg-root} + D-\text{int}_{aqveg} \times RBE \qquad \{\text{roots}\}$$

where:

$D\text{-}ext_{aqveg\text{-}leaf}$ = external dose to aquatic vegetation leaves reported as milliGray per day [mGy/d]

$D\text{-}ext_{aqveg\text{-}root}$ = external dose to aquatic vegetation roots reported as milliGray per day [mGy/d]

$D\text{-}int_{aqveg}$ = internal dose to aquatic vegetation reported as milliGray per day [mGy/d]

RBE = relative biological effectiveness factor

$D\text{-}eq_{aqveg\text{-}leaf}$ = equivalent dose to aquatic vegetation leaves reported as milliGray per day [mGy/d]

D-eqaqveg-root = equivalent dose to aquatic vegetation roots reported as milliGray per day [mGy/d]

2.8.3 Benthic Invertebrates

In the INTAKE model, the concentration in benthic invertebrates is calculated as a function of the transfer factor and the concentration in the water phase from the relationship shown in equation (2.8-12):

$$C_{benthic} = C_{water} \times TF_{benthic} \qquad (2.8\text{-}12)$$

where:

$C_{benthic}$ = concentration of constituent in benthic invertebrates reported as Becquerels per gram (wet weight) or grams per gram (wet weight) [Bq/g (wet weight) or g/g (wet weight)]

C_{water} = water concentration reported as Becquerels per cubic meter or grams per cubic meter [Bq/m^3 or g/m^3]

$TF_{benthic}$ = the water-to-benthic invertebrates transfer factor of radionuclide or metal (Bq/g (wet weight) per Bq/m^3) or (g/g (wet weight) per g/m^3)

One of the environmental components in the pathways model is insects. Flying insects generally have an aquatic larval stage and a terrestrial adult stage. Among the more important groups with benthic larvae are the *Chironomidae* (midges), *Simuliidae* (black flies), *Ephemeroptera* (mayflies), *Plecoptera* (stoneflies), *Odonata* (dragonflies and damselflies) and *Trichoptera* (caddisflies). The pathways model considers flying insects, which are modelled as an insect with an aquatic benthic larval phase and assumes that the insect emerges and becomes a flying adult invertebrate with the same body concentration as in the aquatic larval phase.

Transfer factors for benthic invertebrates are provided in Section 5.1.2.3. For the assessment of dose from radionuclides, concentrations in benthic invertebrates are predicted as described in

equation 2.8-12 and this concentration is converted into an internal dose as shown in equation 2.8-13:

$$D-\text{int}_{benthic} = C_{benthic} \times DC_{\text{int}} \quad \text{(2.8-13)}$$

where:
$D\text{-int}_{benthic}$ = internal dose to benthic invertebrates reported as milliGray per day [mGy/d]
$C_{benthic}$ = concentration of constituent in benthic invertebrates reported as Becquerels per gram (wet weight) or grams per gram (wet weight) [Bq/g (wet weight) or g/g (wet weight)] {equation 2.8-12}
DC_{int} = internal dose coefficient reported as milliGray per day per Becquerels per gram (wet weight) [mGy/d per Bq/g (wet weight)]

Internal dose coefficients for ecological receptors are provided in Section 5.1.4.2.

The assessment of dose from radionuclides for benthic invertebrates also considers the external dose from sediment. The external dose is calculated as shown in equation 2.8-14:

$$D-ext_{benthic} = C_{sed} \times DC_{ext} \quad \text{(2.8-14)}$$

where:
$D\text{-ext}_{benthic}$ = external dose to benthic invertebrates reported as milliGray per day [mGy/d]
C_{sed} = sediment concentration reported as Becquerels per grams [Bq/g]
DC_{ext} = external dose coefficient reported as milliGray per day per Becquerels per gram (wet weight) [mGy/d per Bq/g (wet weight)]

External dose coefficients for ecological receptors are provided in Section 5.1.4.2.

The total absorbed radiological dose for benthic invertebrates is calculated as the sum of the internal and external doses, as shown in equation 2.8-15:

$$D-tot_{benthic} = D-ext_{benthic} + D-\text{int}_{benthic} \quad \text{(2.8-15)}$$

where:
$D\text{-ext}_{benthic}$ = external dose to benthic invertebrates reported as milliGray per day [mGy/d]
$D\text{-int}_{benthic}$ = internal dose to benthic invertebrates reported as milliGray per day [mGy/d]
$D\text{-tot}_{benthic}$ = total absorbed dose to benthic invertebrates reported as milliGray per day [mGy/d]

For the evaluation of radiological dose to benthic invertebrates, the absorbed dose is multiplied by an appropriate radiation weighting factor to account for differences in the effects of alpha, beta and gamma radiation. This is achieved through the use of a relative biological effectiveness (RBE) factor (see Section 6.5.1), which is applied to the absorbed dose to calculate an equivalent dose, as shown in equation 2.8-16:

$$D-eq_{benthic} = D-ext_{benthic} + D-\text{int}_{benthic} \times RBE \qquad (2.8\text{-}16)$$

where:
$D\text{-}ext_{benthic}$ = external dose to benthic invertebrates reported as milliGray per day [mGy/d]
$D\text{-}int_{benthic}$ = internal dose to benthic invertebrates reported as milliGray per day [mGy/d]
RBE = relative biological effectiveness factor
$D\text{-}eq_{benthic}$ = equivalent dose to benthic invertebrates reported as milliGray per day [mGy/d]

2.8.4 Phytoplankton and Zooplankton

The concentration in zooplankton and phytoplankton is calculated as a function of the transfer factor and the concentration in the water phase from the relationship shown in equation (2.8-17):

$$C_{plankton} = C_{water} \times TF_{plankton} \qquad (2.8\text{-}17)$$

where:
$C_{plankton}$ = concentration of constituent in zooplankton or phytoplankton reported as Becquerels per gram (wet weight) [Bq/g (wet weight)]
C_{water} = water concentration reported as Becquerels per cubic meter [Bq/m^3]
$TF_{plankton}$ = the water-to-plankton transfer factor of radionuclide (Bq/g (wet weight) per Bq/m^3)

Transfer factors for zooplankton and phytoplankton are provided in Section 5.1.2.4. For the assessment of dose from radionuclides, this concentration is converted into an internal dose as shown in equation 2.8-18:

$$D-\text{int}_{plankton} = C_{plankton} \times DC_{\text{int}} \qquad (2.8\text{-}18)$$

where:
$D\text{-}int_{plankton}$ = internal dose to zooplankton or phytoplankton reported as milliGray per day [mGy/d]
$C_{plankton}$ = concentration of constituent in zooplankton or phytoplankton reported as Becquerels per gram (wet weight) [Bq/g (wet weight)] {equation 2.8-17}
DC_{int} = internal dose coefficient reported as milliGray per day per Becquerels per gram (wet weight) [mGy/d per Bq/g (wet weight)]

Internal dose coefficients for ecological receptors are provided in Section 5.1.4.2.

The assessment of dose from radionuclides for plankton also considers the external dose from water. The external dose is calculated as shown in equation 2.8-19:

$$D-ext_{plankton} = C_{water} \times DC_{ext} \div 1000000 \qquad (2.8\text{-}19)$$

where:

$D\text{-}ext_{plankton}$ = external dose to zooplankton or phytoplankton reported as milliGray per day [mGy/d]

C_{water} = water concentration reported as Becquerels per cubic meter or grams per cubic meter [Bq/m^3 or g/m^3]

1000000 = unit conversion factor for m^3 to g

DC_{ext} = external dose coefficient reported as milliGray per day per Becquerels per gram (wet weight) [mGy/d per Bq/g (wet weight)]

External dose coefficients for ecological receptors are provided in Section 5.1.4.2.

The total absorbed radiological dose for zooplankton and phytoplankton is calculated as the sum of the internal and external doses, as shown in equation 2.8-20:

$$D-tot_{plankton} = D-ext_{plankton} + D-\text{int}_{plankton} \qquad (2.8\text{-}20)$$

where:

$D\text{-}ext_{plankton}$ = external dose to zooplankton and phytoplankton reported as milliGray per day [mGy/d]

$D\text{-}int_{plankton}$ = internal dose to zooplankton and phytoplankton reported as milliGray per day [mGy/d]

$D\text{-}tot_{plankton}$ = total absorbed dose to zooplankton and phytoplankton reported as milliGray per day [mGy/d]

For the evaluation of radiological dose to zooplankton and phytoplankton, the absorbed dose is multiplied by an appropriate radiation weighting factor to account for differences in the effects of alpha, beta and gamma radiation. This is achieved through the use of a relative biological effectiveness (RBE) factor (see Section 6.5.1), which is applied to the absorbed dose to calculate an equivalent dose, as shown in equation 2.8-21:

$$D-eq_{plankton} = D-ext_{plankton} + D-\text{int}_{plankton} \times RBE \qquad (2.8\text{-}21)$$

where:

$D\text{-}ext_{plankton}$ = external dose to zooplankton and phytoplankton reported as milliGray per day [mGy/d]

$D\text{-}int_{plankton}$ = internal dose to zooplankton and phytoplankton reported as milliGray per day [mGy/d]

RBE = relative biological effectiveness factor

$D\text{-}eq_{plankton}$ = equivalent dose to zooplankton and phytoplankton reported as milliGray per day [mGy/d]

2.9 Terrestrial Biota Model

Modelling the uptake of radionuclides and metals by terrestrial biota can be quite involved and requires consideration of several interactions. For example, direct deposition from air to soil and terrestrial vegetation, direct uptake from soil to terrestrial vegetation, and ingestion of diet components by terrestrial animals and birds are considered in the modelling. The bioaccumulation factors used in the terrestrial biota model are an aggregate representation of numerous pathways and mechanisms of radionuclide and metal transfer through the environment.

2.9.1 Soil

The soil model simulates the movement in soil of radionuclides and metals that are deposited as particulate matter from the air. The soil model includes atmospheric deposition, surface runoff, downward transport of metals and radionuclides into the soil profile, and radioactive decay.

Activity in the soil is calculated and is subsequently used to compute groundshine dose. The radioactivity and metal concentrations may also be used to predict the dose due to consumption of vegetation grown in the soil via root uptake and due to the inadvertent ingestion of soil by animals or people.

To simplify the modelling exercise, the following depletion mechanisms are not included in the model: dust resuspension; plume depletion due to vegetation uptake; and soil erosion. In terms of the total amounts of constituents in the soil, the above mechanisms are relatively insignificant. However, the exclusion of these removal mechanisms leads to a conservative overestimate of the level of constituents in the soil.

The major assumption used in the soil model is that incoming radioactivity and metals are instantaneously partitioned between the liquid and solid phases according to parameter specific distribution coefficients (kd values). The activity or mass balance in the surface soil layer (referred to in the soil model as layer 1) receiving radioactivity or metals deposited from the air can be mathematically formulated (approximately) as equation (2.9-1):

$$dZ_1 = C_a V_d A dt - C_w P A dt - \lambda Z_1 dt \quad (2.9\text{-}1)$$

where:

Z_1 = constituent content in soil layer 1 (Bq or g)

A = soil area (m^2) [taken as 1 m^2 in model calculations]

C_a = airborne particle concentration reported as Becquerels per cubic meter or grams per cubic meter [Bq/m^3 or g/m^3]

V_d = particle deposition velocity (m/y)

t = time (yr)

C_w = concentration in soil water reported as Becquerels per cubic meter or grams per cubic meter [Bq/m^3 or g/m^3]

P = total precipitation less evapotranspiration (i.e., net precipitation) [m/y]

λ = radioactive decay constant (1/y)

The first term on the right hand side of the equation (2.9-1) accounts for deposition on soil from air. The second term accounts for removal via leaching in both surface and subsurface flow. The third term accounts for radioactive decay. The radionuclide and metal concentrations in the subsurface layer, which is modelled as the root zone from which constituents are taken up by vegetation, are estimated using a similar formulation as described above.

Soil model default input parameters are summarized in Table 2.9-1.

Table 2.9-1 Soil Model Input Parameters

Parameter Description	Units	Default Parameter Distribution	Reference
deposition velocity	m/s	U(0.0005, 0.005)	Based on results of air dispersion modelling
depth of soil layer k			
- Layer 1	m	C(0.08)	SENES 1987
- Layer 2	m	C(0.20)	SENES 1987
fraction of net precipitation in runoff	-	U (0.02, 0.51)	Assumed
soil bulk density of kth soil layer			
- Layer 1	kg/m^3	LN(129, 1.3, 59, 280)	SENES 1987
- Layer 2	kg/m^3	U(800, 1000)	Beak 1987
Water content of kth soil layer	kg (water) per kg (dry soil)		
- Layer 1		U(0.1, 0.4)	Beak 1987
- Layer 2		U(0.1, 0.4)	Beak 1987

2.9.2 Groundshine

The groundshine model calculates the receptor dose due to surface gamma flux (groundshine) attributed to the accumulation over time of selected radionuclides in the soil profile. The radionuclides considered in the implementation of the groundshine model include the major gamma emitting radionuclides in the uranium-238 decay series. The model uses the radionuclide concentrations in the soil layers that are calculated in the soil model. The groundshine model accounts for self-shielding by overlying soil layers and the loss of gamma ray flux via radon emanation and subsequent exhalation to the atmosphere.

In the development of the groundshine model, a number of assumptions have been introduced to simplify the computations. While the assumptions are discussed in the Component Model Documentation for UTAP Version 3 (SENES 1987), two assumptions are worth noting here.

First, any attenuation which may take place in the medium (air) between the source (i.e., the soil layer) and the receptor is neglected as is any attenuation or scatter which might be expected to occur in the thin overlying organic soil layer. Second, a simplifying approximation has been to use the linear absorption coefficient rather than the linear attenuation coefficient in estimating the gamma ray flux from the soil. This assumption more than compensates for the effect of scattered radiation originating from within the self-absorbing source material (i.e., the soil) and results in conservative overestimates of gamma ray exposure.

2.9.3 Terrestrial Vegetation

In the terrestrial vegetation model, air and soil concentrations are used to model the concentration of a radionuclide or metal in six classes of terrestrial vegetation: above ground, leafy vegetables; below ground vegetables; summer forage; browse; lichen; and berries.

One basic model is used for all types of terrestrial vegetation, with modifications for lichen. Airborne materials can be transferred to the edible portions of fruits and vegetables consumed by people or to forage consumed by animals either by direct foliar retention or root uptake. The following equation was adapted from work presented by the US NRC (1982) to calculate the average annual concentration of each radionuclide or metal in each vegetation type (equation 2.9-2):

$$C_v = C_a V_d F_{in} F_{rv} E_v \left[\frac{1 - \exp(\lambda_{wv} t_v)}{Y_v \lambda_{wv}} \right] + \frac{B_v C_g}{\rho_g} \quad \text{(2.9-2)}$$

where:

C_v = constituent concentration in each vegetation type (Bq/g (wet weight) or g/g (wet weight))

C_a = airborne particle concentration reported as Becquerels per cubic meter or grams per cubic meter [Bq/m^3 or g/m^3]

V_d = particle deposition velocity (m/yr)
F_{in} = fraction of deposition that is intercepted by each specific vegetation type (0 to 1)
F_{rv} = fraction of deposition that is retained on the plant surface of each specific vegetation type (0 to 1)
E_v = fraction of foliar deposition that is retained on the edible portions of each specific vegetation type (0 to 1)
λ_{wv} = a decay constant accounting for weathering losses from each specific vegetation type (1/sec) and if significant, radioactive decay
t_v = duration of plant exposure to atmospheric deposition (sec)
Y_v = yield density for each specific vegetation type (g (wet weight)/m^2)
B_v = soil-to-plant transfer coefficient for each radionuclide or constituent and each specific vegetation type (Bq/g (wet weight) plant per Bq/g (dry weight) soil or g/g(wet weight) plant per g/g (dry weight) soil)
C_g = the average concentration of each radionuclide or constituent in the root zone of the soil (Bq/m^3 or g/m^3) {calculated in the soil model}
ρ_g = density of soil (g (dry weight)/m^3)

The first term on the right hand side of the equation represents direct foliar retention; the second term represents root uptake. Root uptake is considered to be insignificant for lichen (due to the limited root structure) and berries (negligible translocation to berries from root uptake). Irrigation is not included in this application. Note also that time dependence is due only to the time dependence of the air and soil concentrations and vegetation is calculated based only on the conditions at each time step.

The model assumes that the system is in equilibrium. For the foliar deposition model, this means that the weathering half-life is sufficiently less than the period of deposition and radioactive half-life of the radionuclides. Therefore the concentration of each radionuclide (or metal) reaches a steady state level. For vegetation that has a lifespan of one growing season this is a valid assumption.

The model is modified for the estimation of concentrations in lichen to incorporate a COPC-specific component to represent uptake, retention, and subsequent release (equation 2.9-3).

$$\Delta C_v = \Delta C_a V_d F_{in} F_{rv} E_v \left[\frac{1 - \exp(\lambda_{wv} t_v)}{RTY_v \lambda_l} \right] \qquad (2.9\text{-}3)$$

where:
ΔC_v = incremental constituent concentration in lichen (Bq/g (wet weight) or g/g (wet weight))
ΔC_a = incremental airborne particle concentration reported as Becquerels per cubic meter or grams per cubic meter [Bq/m^3 or g/m^3]
V_d = particle deposition velocity (m/yr)
F_{in} = fraction of deposition that is intercepted by each specific vegetation type (0 to 1)

F_{rv} = fraction of deposition that is retained on the plant surface of each specific vegetation type (0 to 1)

E_v = fraction of foliar deposition that is retained on the edible portions of each specific vegetation type (0 to 1)

λ_{wv} = a decay constant accounting for weathering losses from each specific vegetation type (1/sec) and if significant, radioactive decay

t_v = duration of plant exposure to atmospheric deposition (sec)

Y_v = yield density for each specific vegetation type (g (wet weight)/m^2)

RT = residence time (assumed to be 5)

λ_l = a constituent-specific decay constant to represent uptake, retention, and release as well as weathering loss and if significant, radioactive decay (sec)

There is no transfer from soil to lichen considered in the model. According to Walther et al. (1990), the biological mean residence time is 2-5 years for most elements in lichens. As a simplification, lichens were taken to be in equilibrium with the air within one year. The model does not allow lichen concentrations to fall below a minimum concentration associated with background air concentrations.

Although there is a focus in literature on the study of lichen as biomonitors from the point of view of accumulation, there is some work related to elemental leaching or release. Wolterbeek (2003) examined the dynamics of metal uptake and release and reports on several studies that illustrate the release of contaminants, such as lead and uranium, from lichen once they are returned to ambient/unaffected conditions, in a matter of months. Walther et al. (1990) notes that although lichens accumulate at a rapid rate, they release metals at a slower rate.

According to Walther et al. (1990), the biological mean residence time is two to five years for most elements in lichens. Similar results were reported by Nieboer et al. (1978) in a review of the mineral uptake and release by lichens. It was found that the measured mean residence or turnover times are reported as two to seven years for ions accumulated both extra- and intra-cellularly. The natural depuration of Cs-137 in lichen was studied and it was discovered that the biological half-life in *Xanthoria parietina* lichen is 58.6 months (Topcuoglu et al. 1995). Lichens are known to participate in mineral cycling in the environment, and as part of their role in mineral cycling and returning stored elements to the environment, lichen have the appropriate mechanisms to release stored mineral substances.

Fahselt et al. (1995) measured trace elements in lichen along transects extending from uranium mines at Elliot Lake and Agnew Lake in central Ontario, Canada. Agnew Lake mine closed in 1983 and Quirke Number 2 at Elliot Lake ceased production in 1992. Lichen samples were collected in 1992 and compared to data collected during the operation phase in 1982. They found that although the presence of uranium was confirmed the concentrations were much less in the previous samples. Similarly, Trembley et al. (1997) reported a substantial decrease in levels of uranium in lichen near Elliot Lake, Ontario within three years of ceased uranium mining operations. The uptake of uranium

by lichen is mainly from airborne particulates and dust and therefore, the release of uranium from lichen is largely attributable to the physical detachment of particulates from the lichen, but could also be due to precipitation that can displace elements held within a lichen thallus.

Vegetation model default input parameters are summarized in Table 2.9-2. Transfer factors are defined in Section5.1.

Table 2.9-2 Terrestrial Vegetation Model Input Parameters

Parameter Description	Units	Parameter Distribution	Reference
fraction of total deposition retained on plant surface (F_{rv})			
- summer forage	-	U(0.1, 0.3)	SENES 1987
- browse	-	U(0.1, 0.3)	SENES 1987
- lichen	-	U(0.8, 1.0)	SENES 1987
- berries	-	U(0.1, 0.3)	Beak 1987
fraction of plant deposition retained on edible portions (E_v)			
- summer forage	-	U(0.8, 1.0)	SENES 1987
- browse	-	U(0.8, 1.0)	SENES 1987
- lichen	-	U(0.8, 1.0)	SENES 1987
- berries	-	U(0.8, 1.0)	Beak 1987
duration of exposure to deposition (t_v)			
- summer forage	s	U(1.3×10^{6}, 3.9×10^{6})	SENES 1987
- browse	s	U(6.8×10^{6}, 8.8×10^{6})	Beak 1985
- lichen	s	C(1×10^{10})	SENES 1987
- berries	s	U(6.8×10^{6}, 8.8×10^{6})	Beak 1987
yield density (Y_v)			
- summer forage	g(wet)/ m^2	U(5.0×10^{2}, 1.0×10^{3})	SENES 1987
- browse	g(wet)/ m^2	U(5.0×10^{2}, 1.0×10^{3})	SENES 1987
- lichen	g(wet)/ m^2	U(4.0×10^{2}, 6.0×10^{2})	SENES 1987
- berries	g(wet)/ m^2	U(75, 125)	Beak 1987
decay constant accounting for weathering loss (λ_{wv})			
- summer forage	1/s	U(2.9×10^{-7}, 8.6×10^{-7})	SENES 1987
- browse	1/s	U(2.9×10^{-7}, 8.6×10^{-7})	SENES 1987
- lichen*	1/s	U(6.6×10^{-8}, 1.9×10^{-7})	Adjusted
- berries	1/s	U(2.9×10^{-7}, 8.6×10^{-7})	Beak 1987
Note: * λ_{wv} used as a calibration variable for lichen to ensure predicted operational values reflect measured values.			

2.9.4 Animals

The terrestrial biota model draws on information from several other models to calculate the concentration of radionuclides and metals in animal flesh. The specific receptors considered for the assessment are characterized in Section 4, but the general model equations are provided below by animal diet-type.

2.9.4.1 Herbivores

For herbivores, the calculation of the concentration of each radionuclide and metal may include many possible subsets of browse, forage, berries, lichen, soil, water, benthic invertebrates, sediment or aquatic vegetation-to-herbivore flesh pathways. Radionuclides and metals in terrestrial and aquatic vegetation and water may be eaten by herbivores, which may in turn be eaten by omnivores, carnivores or people. This transfer for each constituent is modelled as follows in equation (2.9-4):

$$C_h = \left(\sum \left(C_v F_{h,v}\right) Q_h + C_w Q_{h,w}\right) T_h \qquad (2.9\text{-}4)$$

where:

C_h = annual average concentration of constituent in herbivore (type "h") flesh (Bq/g (wet weight) or g/g (wet weight))

C_v = average concentration of each constituent in vegetation type "v" (Bq/g (wet weight) or g/g (wet weight))

$F_{h,v}$ = fraction of feed for herbivore type "h" that is vegetation type "v"

Q_h = average daily feed intake of herbivore type "h" (g (wet weight)/d)

C_w = average concentration of each constituent in water (Bq/m^3 or g/m^3)

$Q_{h,w}$ = average daily water intake of herbivore type "h" (m^3/d)

T_h = feed-to-herbivore (type "h") flesh transfer coefficient (Bq/g (wet weight) per Bq/d or g/g (wet weight) per g/d)

2.9.4.2 Omnivores

In addition to vegetation and drinking water, omnivores may consume aquatic and/or terrestrial herbivores, omnivores or carnivores. The transfer for each constituent to omnivores is modelled as follows in equation (2.9-5):

$$C_o = \left(\sum \left(C_v F_{o,v}\right) Q_o + \sum \left(C_m F_{o,m}\right) Q_o + C_w Q_{o,w}\right) T_o \qquad (2.9\text{-}5)$$

where:

C_o = annual average concentration of constituent in omnivore (type "o") flesh (Bq/g (wet weight) or g/g (wet weight))

C_v = average concentration of each constituent in vegetation type “v” (Bq/g (wet weight) or g/g (wet weight))
$F_{o,v}$ = fraction of feed for omnivore type “o” that is vegetation type “v”
Q_o = average daily feed intake of omnivore type “o” (g (wet weight)/d)
C_m = average concentration of each constituent in meat type “m” (Bq/g (wet weight) or g/g (wet weight))
$F_{o,m}$ = fraction of feed for omnivore type “o” that is meat type “m”
C_w = average concentration of each constituent in water (Bq/m^3 or g/m^3)
$Q_{o,w}$ = average daily water intake of omnivore type “o” (m^3/d)
T_o = feed-to-omnivore (type “h”) flesh transfer coefficient (Bq/g (wet weight) per Bq/d or g/g (wet weight) per g/d)

2.9.4.3 Carnivores

Carnivores consume drinking water and aquatic and/or terrestrial herbivores, omnivores or carnivores. The transfer for each constituent to carnivores is modelled as follows in equation (2.9-6):

$$C_c = \left(\sum \left(C_m F_{c,m}\right) Q_c + C_w Q_{c,w}\right) T_c \qquad (2.9\text{-}6)$$

where:
C_c = annual average concentration of constituent in carnivore (type “c”) flesh (Bq/g (wet weight) or g/g (wet weight))
C_m = average concentration of each constituent in meat type “m” (Bq/g (wet weight) or g/g (wet weight))
$F_{c,m}$ = fraction of feed for carnivore type “c” that is meat type “m”
Q_c = average daily feed intake of carnivore type “c” (g (wet weight)/d)
C_w = average concentration of each constituent in water (Bq/m^3 or g/m^3)
$Q_{c,w}$ = average daily water intake of carnivore type “c” (m^3/d)

2.9.4.4 Radiological Dose

For the assessment of radiological dose to animals, calculated concentrations in herbivores, omnivores, and carnivores (see Sections 2.9.4.1 to 2.9.4.3, respectively) were used to estimate an internal dose from radionuclides, as shown in equation (2.9-7):

$$D_{\text{int}} = C \times DC_{\text{int}} \qquad (2.9\text{-}7)$$

where:
C = annual average concentration of radionuclide in herbivore, omnivore, and carnivore flesh (Bq/g (wet weight))
DC_{int} = internal dose coefficient (mGy/d per Bq/g)
D_{int} = internal dose (mGy/d)

Internal dose coefficients for ecological receptors are provided in Section 5.1.4.2.

The assessment of dose from radionuclides for animals also considers the external dose from gamma. The external dose is calculated as shown in equation 2.9-8:

$$D_{ext} = \gamma_{rate} \times floc \times 24 \div 1000 \qquad (2.9\text{-}8)$$

where:
γ_{rate} = exeternal gamma rate (uGy/hr)
floc = fraction of time on site (-)
24 = conversion factor (hrs/d)
1000 = conversion factor (uGy/mGy)
D_{ext} = external dose from gamma (mGy/d)

The total absorbed dose from radiological constituents for animals was calculated as the sum of the internal and external doses, as shown in equation (2.9-9):

$$D_{total} = D_{ext} + D_{\text{int}} \qquad (2.9\text{-}9)$$

where:
D_{ext} = external dose to animal reported as milliGray per day [mGy/d]
D_{int} = internal dose to animal reported as milliGray per day [mGy/d]
D_{total} = total absorbed dose to animal reported as milliGray per day [mGy/d]

For the evaluation of radiological dose to animals, the absorbed dose is multiplied by an appropriate radiation weighting factor to account for differences in the effects of alpha, beta and gamma radiation. This is achieved through the use of a relative biological effectiveness (RBE) factor (see Section 6.5.1), which is applied to the absorbed dose to calculate an equivalent dose, as shown in equation 2.9-10:

$$D - eq = D_{ext} + D_{\text{int}} \times RBE \qquad (2.9\text{-}10)$$

where:
D_{ext} = external dose to animals reported as milliGray per day [mGy/d]
D_{int} = internal dose to animals reported as milliGray per day [mGy/d]
RBE = relative biological effectiveness factor
D-eq = equivalent dose to animals reported as milliGray per day [mGy/d]

2.10 Human Models

The human model is comprised of the inhalation and ingestion models. Airborne dust and radon account for the inhalation dose while consumption of water, fish, vegetables, fruits, and wildlife account for the ingestion dose. In addition, an external dose is measured due to gamma radiation from the radionuclide content of dust deposited on the ground.

2.10.1 Ingestion Model

The annual average intake of radionuclides and metals by an individual due to the ingestion of above ground and below ground vegetables and fruit (berries) was calculated as follows in equation (2.10-1):

$$I_v = \sum_v U_v C_v R_v L_v \qquad (2.10\text{-}1)$$

where:
I_v = annual intake of each constituent due to ingestion of vegetation (Bq/year or g/year)
U_v = average annual intake of each vegetation type (g (wet weight)/year)
C_v = calculated average annual concentration of each constituent in each vegetation type "v" (Bq/g (wet weight) or g/g (wet weight))
R_v = fraction of each constituent in each vegetation type remaining after food preparation (0 to 1)
L_v = fraction of each vegetation type that is grown locally (0 to 1)

The average annual intake of radionuclides and metals by an individual due to drinking water was modelled (NUREG 1982) as shown in equation (2.10-2):

$$I_w = U_w \sum_w C_w L_w \qquad (2.10\text{-}2)$$

where:
I_w = annual intake of each constituent due to ingestion of drinking water (Bq/year or g/year)
U_w = average annual intake of drinking water (m^3/year)
C_w = calculated average annual concentration of each constituent in drinking water from each source "w" (Bq/m^3 or g/m^3)

L_w = fraction of the annual water from each drinking water source “w” (0 to 1)

The annual average intake of radionuclides and metals by an individual due to the ingestion of fish and meat was calculated as follows in equation (2.10-3):

$$I_f = \sum_f U_f C_f L_f \qquad (2.10\text{-}3)$$

where:
I_f = annual intake of each constituent due to ingestion of fish and meat (food type “f”) (Bq/year or g/year)
U_f = average annual intake of each food type “f” (g (wet weight)/year)
C_f = calculated average annual concentration of each constituent in each food type “f” (Bq/g (wet weight) or g/g (wet weight))
L_f = fraction of each food type that is taken locally (0 to 1)

The annual average intake of radionuclides and metals by an individual due to the ingestion of soil was calculated as follows in equation (2.10-4):

$$I_s = \sum_s U_s C_s L_s \qquad (2.10\text{-}4)$$

where:
I_s = annual intake of each constituent due to ingestion of soil (Bq/year or g/year)
U_s = average annual intake of soil (g/year)
C_s = calculated average annual concentration of each constituent in soil at location “s” (Bq/g (wet weight) or g/g (wet weight))
L_s = fraction of time spent at each soil location “s” (0 to 1)

2.10.2 Inhalation Model

The average annual intake by an individual due to inhalation (excluding radon) is calculated as follows in equation (2.10-5):

$$I_i = U_i C_a \qquad (2.10\text{-}5)$$

where:
I_i = annual intake of each constituent due to inhalation (Bq/year or g/year)
U_i = average annual inhalation rate (m^3/year)
C_a = calculated average annual concentration of each constituent in air (Bq/m^3 or g/m^3)

The average annual dose of radon by inhalation is calculated as follows in equation (2.10-6):

$$D_{Rn} = C_{Rn} \times OF \times INOCC \times F \times 69.6 \quad \text{(2.10-6)}$$

where:
D_{Rn} = annual dose of radon and progeny by inhalation (µSv/year)
C_a = average annual concentration of radon in air (Bq/m^3)
OF = fraction of year at location (-)
INOCC = fraction of time spent indoors (-)
F = equilibrium factor between radon and its progeny
69.6 = conversion factor from Bq/m^3 to µSv/year

Since it is the radon progeny rather than the radon that produces the dose to the receptor, the radon-in-air concentration must be converted to an appropriate radon progeny concentration. This conversion requires an estimation of the equilibrium factor, F, between radon and its progeny. Theoretically, F can range from 0.0 (no radon progeny) to 1.0 (complete radon progeny ingrowth). The value of F outdoors at the receptor depends only on the transit time between the radon source and the receptor if it is assumed that there are no radon progeny removal mechanisms other than radioactive decay. Since in reality the progeny are removed by several other mechanisms that are difficult to quantify (e.g., attachment to dust that falls to the ground) and since the major exposure occurs indoors where F is dependent on several additional factors (such as the building ventilation rate), it has been customary to select an F factor that is representative of annual average indoor conditions. The radon concentration at the receptor is converted to a radon progeny concentration using this F factor. For this study, the assumed F value is selected from an input distribution, triangular in shape, with a range of 0.2 to 0.6 and a mode of 0.4. The equilibrium factor is based on a report by U.S. NRC (1991) which states that the equilibrium factor in houses without smokers is in the range of 0.3 to 0.4.

ICRP Publication No. 65 (1993) suggests, based on equality of detriment, that 1 WLM (Working Level Month) corresponds to (about) 5 mSv for workers and 4 mSv for members of the public (para 56). The formula for estimating effective dose given in the Radiation Protection Regulations of the Canadian Nuclear Safety Commission infer a conversion factor of 1 WLM to 5 mSv for a member of the public. This latter factor is used for this study.

2.10.3 Groundshine Model

The average annual dose from groundshine (external dose) is calculated as follows in equation (2.10-7):

$$D_{ext} = D \times OF \times \left(\left(1 - INOCC\right) + INOCC \times SHLD \right) \quad \text{(2.10-7)}$$

where:
D_{ext} = annual dose due to groundshine of radionuclide (µSv/year)
D = average groundshine dose rate due to each radionuclide (µSv/yr)
OF = fraction of year at location (-)
INOCC = fraction of time spent indoors (-)
SHLD = effective reduction factor for structural shielding for indoor exposure periods

The fraction of time indoors (INOCC) for adults, children, and toddler was assumed to range on a triangular distribution of T(0.33, 0.5, 0.6). For a hunter, the fraction of time indoors while on the trapline was assumed to range on a triangular distribution of (0.2, 0.3, 0.4); while not on the trapline, the fraction of time indoors for a trapper was considered on a triangular distribution of T(0.33, 0.5, 0.6).

When indoors, protection afforded by buildings from external gamma radiation varies considerably. According to the U.S. NCRP (1984), reported reduction factors (ratio of the exposure rate in the structure to the exposure rate in open air) for radionuclides deposited on the ground range from an average of about 0.27 for single story wooden houses to 0.06 for reinforced concrete and brick homes. Basements provide more protection than other areas of the house and average reduction factors range from 0.003 to 0.08. The U.S. Nuclear Regulatory Commission (NUREG 1992) provides a shielding factor of 0.33 for a residential scenario. This was assumed to vary on a triangular distributio of T(0.15, 0.33, 0.51).

The fraction of year at the site is a function of the human receptor characteristics used in the assessment and described in Section 4.2.1.4

2.10.4 Total Intake

The annual average total intake of radionuclides and metals by an individual was calculated as follows in equation (2.10-8):

$$I_{total} = \sum_{v} I_v + \sum_{w} I_w + \sum_{f} I_f + \sum_{s} I_s + I_i \quad \text{(2.10-8)}$$

where:
I_{total} = total annual intake of each constituent by an individual (Bq/year or g/year)
I_v = annual intake of each constituent due to ingestion of vegetation (Bq/year or g/year)

I_w = annual intake of each constituent due to ingestion of drinking water (Bq/year or g/year)
I_f = annual intake of each constituent due to ingestion of fish and meat (food type "f") (Bq/year or g/year)
I_s = annual intake of each constituent due to ingestion of soil (Bq/year or g/year)
I_i = annual intake of each constituent due to inhalation (Bq/year or g/year)

The total intake of radionuclides is converted to a dose and then considered with the calculated doses from radon and groundshine.

.

3 Site Characterization

This section summarizes information on the baseline conditions for the area of the Kiggavik Project used in the assessment. Site-specific data were augmented as necessary with data from other locations to provide a complete set of baseline (reference) data for incorporation into the pathways analysis of total levels to predict total exposure to the various VECs (broadly, the ERA and HHRA). The assessment presented in this report considers both baseline conditions and the incremental effects of the Kiggavik Project on the atmospheric, aquatic, and terrestrial environments.

3.1 Study Area

Seven areas were defined for the assessment of impacts of the project's atmospheric emissions and liquid effluent releases on the environment. The areas are indicated on Figure 3.1-1 and include:

- Area 1: Rock Lake watershed, not affected by liquid effluent, but closer proximity to atmospheric emissions than other assessment areas outside of the Project site;
- Area 2: Judge Sissons Lake – vicinity of Kiggavik discharge (JSL-2), receiving liquid effluent from the Kiggavik site;
- Area 3: Kiggavik Camp, location of atmospheric emissions
- Area 4: Judge Sissons Lake – vicinity of Sissons discharge (JSL-8), receiving liquid effluent from the Sissons site;
- Area 5: Local Assessment Area, capturing effects from the atmospheric and liquid effluent releases on the local area;
- Area 6: Regional Assessment Area, capturing effects from the atmospheric and liquid effluent releases on the regional area; and,
- Area 7: Baker Lake Area, capturing effects from the atmospheric releases on Baker Lake.

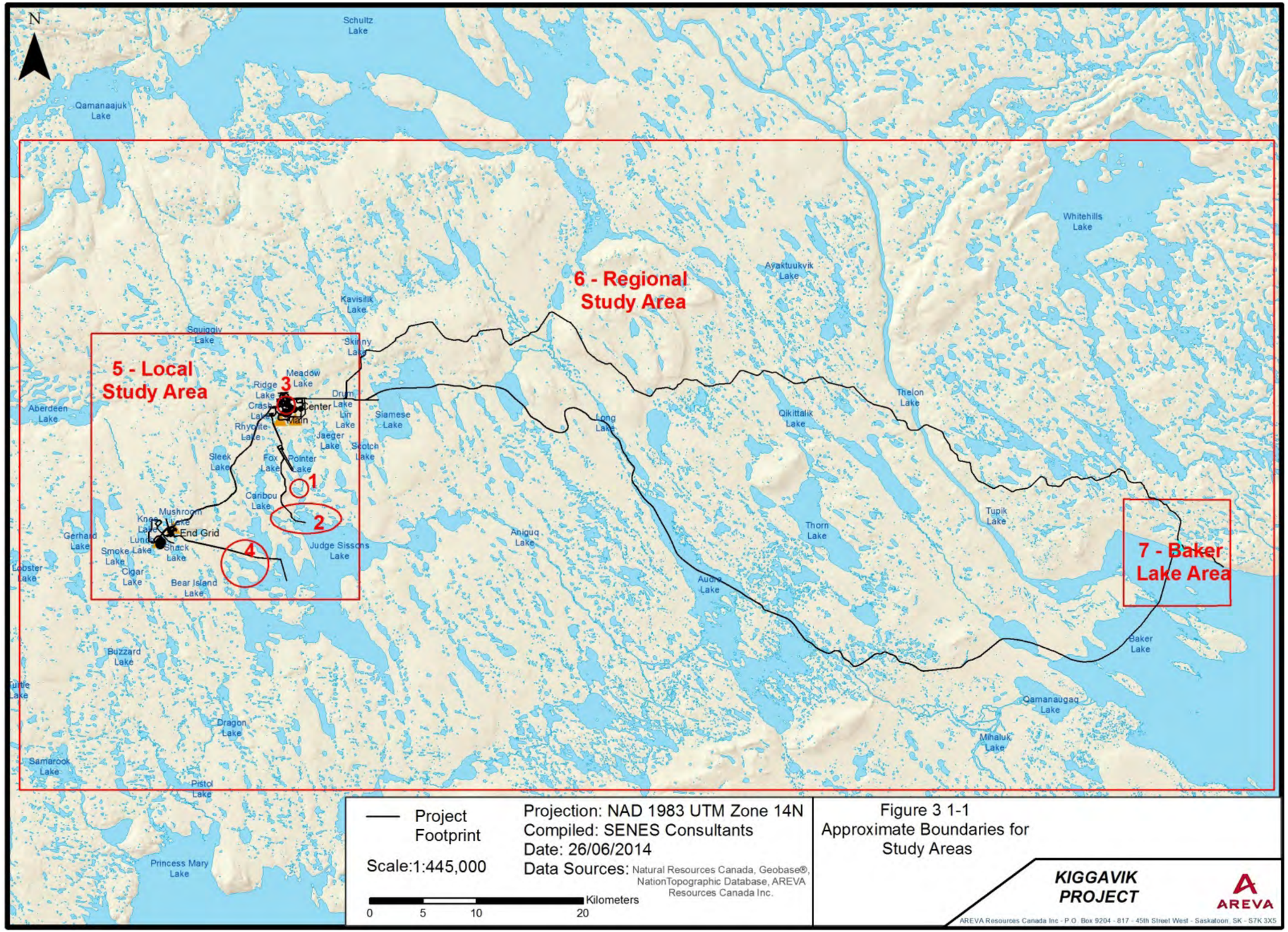


Figure 3.1-1 Assessment Locations within Study Area

3.2 Baseline Constituent Concentrations

Data from monitoring and baseline sampling completed at the site has been summarized for use in the ecological and human health risk assessment. Baseline concentrations of constituents of potential concern (COPC) for the Kiggavik Project area were developed from current data from reference/representative monitoring locations in the study area for air, soil, water, and sediment quality.

For other environmental media, such as terrestrial vegetation, aquatic vegetation, benthic invertebrates, fish, birds, and mammals, concentrations are calculated in the pre-operation phase using transfer factors to represent the transfer of constituents between environmental media. Where available, site-specific data were used to calculate site-specific transfer factors. In the following sections, the calculated concentrations for the pre-operation phase are compared with measured site data, if available, to illustrate the appropriateness of the model estimation.

3.2.1 Baseline Air Quality

The baseline air quality sampling in the local area is described in Appendix 4B. The data collected was used to characterize the baseline air quality for the INTAKE model and is summarized in Table 3.2-1.

Table 3.2-1 Summary of Measured Baseline Air Quality

Air Concentrations											
Constituent	**Units**	**N**	**N<MDL**	**Min**	**Median**	**95th**	**Max**	**Avg**	**St. Dev.**	**GM**	**GSD**
Uranium[a]	µg/m^3	6	5	6.9×10^{-6}	1.0×10^{-5}	1.3×10^{-5}	1.3×10^{-5}	9.9×10^{-6}	2.4×10^{-6}	9.6×10^{-6}	1.3
Thorium-230	Bq/m^3	8	4	1.7×10^{-7}	3.9×10^{-7}	4.3×10^{-6}	5.1×10^{-6}	1.5×10^{-6}	1.8×10^{-6}	6.9×10^{-7}	3.8
Lead-210	Bq/m^3	8	0	8.4×10^{-5}	1.2×10^{-4}	1.7×10^{-4}	1.8×10^{-4}	1.3×10^{-4}	3.7×10^{-5}	1.2×10^{-4}	1.3
Radium-226	Bq/m^3	8	2	1.7×10^{-7}	3.7×10^{-7}	2.7×10^{-6}	2.7×10^{-6}	9.2×10^{-7}	1.1×10^{-6}	5.2×10^{-7}	3.0
Polonium-210	Bq/m^3	8	0	2.7×10^{-5}	4.7×10^{-5}	8.7×10^{-5}	8.8×10^{-5}	5.4×10^{-5}	2.2×10^{-5}	5.0×10^{-5}	1.5
Arsenic	µg/m^3	8	6	3.4×10^{-5}	1.3×10^{-4}	2.2×10^{-4}	2.7×10^{-4}	1.2×10^{-4}	7.2×10^{-5}	1.0×10^{-4}	2.0
Cadmium	µg/m^3	8	3	1.8×10^{-5}	4.6×10^{-5}	1.2×10^{-4}	1.2×10^{-4}	6.0×10^{-5}	3.7×10^{-5}	5.1×10^{-5}	1.9
Cobalt	µg/m^3	8	8	6.9×10^{-6}	4.3×10^{-5}	7.5×10^{-5}	8.9×10^{-5}	4.0×10^{-5}	2.5×10^{-5}	3.2×10^{-5}	2.3
Copper	µg/m^3	8	0	3.4×10^{-3}	1.6×10^{-2}	2.2×10^{-2}	2.4×10^{-2}	1.5×10^{-2}	6.1×10^{-3}	1.4×10^{-2}	1.8
Lead	µg/m^3	8	0	8.2×10^{-5}	7.7×10^{-4}	2.6×10^{-3}	2.7×10^{-3}	1.1×10^{-3}	9.8×10^{-4}	7.3×10^{-4}	3.1
Molybdenum	µg/m^3	8	4	2.6×10^{-5}	1.0×10^{-4}	6.3×10^{-4}	7.3×10^{-4}	2.4×10^{-4}	2.5×10^{-4}	1.4×10^{-4}	3.2
Nickel	µg/m^3	8	0	8.5×10^{-5}	1.6×10^{-4}	2.0×10^{-4}	2.0×10^{-4}	1.5×10^{-4}	4.9×10^{-5}	1.4×10^{-4}	1.4
Selenium[b]	µg/m^3	2	0	2.6×10^{-5}	3.5×10^{-5}	4.3×10^{-5}	4.4×10^{-5}	3.5×10^{-5}	1.2×10^{-5}	3.4×10^{-5}	1.4
Zinc	µg/m^3	8	0	1.6×10^{-3}	4.7×10^{-3}	7.1×10^{-3}	7.3×10^{-3}	4.5×10^{-3}	2.1×10^{-3}	4.0×10^{-3}	1.8

Table 3.2-1 Summary of Measured Baseline Air Quality

Note: Data from sampling (2010 – 2013) described in Appendix 4B Air Quality.
a – data from 2011 for uranium was excluded due to much higher method detection limits than samples collected in other years.
B –data from 2011 – 2013 for selenium was excluded due to much higher method detection limit than samples collected in 2010.

3.2.2 Baseline Soil Quality

Sampling for soil baseline quality was described in Appendix 6B for sampling completed in 2007, 2008, and 2009. Table 3.2-2 provides a summary of the data collected from the site for the COPC. Appendix 6B provides a summary of all of the measured data for soil.

Table 3.2-2 Summary of Measured Baseline Soil Quality

Soil Concentrations - All Locations											
Constituent	**Units**	**N**	**N< MDL**	**Min**	**Median**	**95th**	**Max**	**Avg**	**St. Dev.**	**GM**	**GSD**
Uranium	µg/g dw	16	0	0.7	1.0	2.3	2.4	1.2	0.5	1.1	1.4
Thorium-230	Bq/g dw	46	10	0.010	0.03	0.06	0.13	0.03	0.02	0.03	1.9
Lead-210	Bq/g dw	46	39	0.010	0.01	0.04	0.07	0.02	0.01	0.01	1.7
Radium-226	Bq/g dw	46	4	0.005	0.03	0.07	0.14	0.03	0.02	0.03	2.0
Polonium-210	Bq/g dw	46	2	0.005	0.02	0.04	0.10	0.02	0.01	0.02	1.6
Arsenic	µg/g dw	57	34	2.1	2.5	6.7	13.6	3.8	2.0	3.4	1.5
Cadmium	µg/g dw	66	66	0.05	0.3	0.3	0.3	0.2	0.1	0.2	2.0
Cobalt	µg/g dw	66	0	1.6	4.2	7.1	13.0	4.4	1.9	4.1	1.5
Copper	µg/g dw	66	0	2.0	7.6	18.9	23.0	8.9	5.3	7.5	1.8
Lead	µg/g dw	66	50	4.1	15.0	15.0	23.0	13.7	3.6	13.0	1.5
Molybdenum	µg/g dw	66	48	0.1	2.0	2.0	6.9	1.8	1.2	1.2	2.8
Nickel	µg/g dw	66	1	2.5	11.8	18.7	61.0	12.4	7.7	11.1	1.6
Selenium	µg/g dw	66	65	0.05	1.0	1.5	2.0	0.8	0.5	0.5	3.6
Zinc	µg/g dw	66	0	7.8	16.2	33.7	56.0	18.2	8.2	16.8	1.5

Note:

N = number of samples.

N<MDL = number of samples reported as less than the method detection limit.
Min = minimum; 95th = 95th percentile; Max = maximum; Avg = average.

St.Dev = standard deviation; GM = geometric mean; GSD = geometric standard deviation.

µg/g = micrograms per gram; Bq/g = Bequerels per gram; dw = dry weight basis.

Source: Appendix 6B, data from 2007, 2008, 2009; data reported as less than the method detection limit were treated as equal to ½ the method detection limit

The average values presented in Table 3.2-2 were used to represent the background concentrations for the site.

3.2.3 Baseline Water Quality

Sampling for water baseline quality was described in Appendix 5C for sampling completed in 2007, 2008, and 2009. Table 3.2-3 provides a summary of the data collected from the site for the COPC. Appendix 5C provides a summary of all of the measured data for water.

Additional low level analysis for cadmium was completed in 2013 on samples from Judge Sissons Lake and Squiggly Lakes. The method detection limit achieved for this analysis was 0.000053 µg/L, a summary of the program and results is provided in Attachment 5C-1.

Data for water quality in lakes and rivers were compared and it was determined that water quality was generally comparable between both types of waterbodies and, therefore, data from lakes and rivers were grouped for the summary statistics provided in Table 3.2-3.

Table 3.2-3 Summary of Measured Baseline Water Quality

Water Concentrations - All Locations											
Constituent	**Units**	**N**	**N<MDL**	**Min**	**Median**	**95th**	**Max**	**Avg**	**St. Dev.**	**GM**	**GSD**
Uranium	µg/L	130	114	0.05	0.05	0.2	0.3	0.064	0.043	0.057	1.48
Thorium-230	Bq/L	130	118	0.005	0.005	0.010	0.03	0.006	0.003	0.005	1.33
Lead-210	Bq/L	130	126	0.01	0.01	0.010	0.03	0.011	0.003	0.010	1.19
Radium-226	Bq/L	130	98	0.0025	0.0025	0.009	0.01	0.004	0.002	0.003	1.55
Polonium-210	Bq/L	130	85	0.0025	0.0025	0.010	0.03	0.005	0.005	0.004	1.92
Arsenic	µg/L	130	14	0.05	0.15	0.3	0.5	0.16	0.086	0.14	1.70
Cadmium	µg/L	20	0	0.0010	0.0013	0.0025	0.0026	0.0015	0.0004	0.0014	1.13
Cobalt	mg/L	130	98	0.00005	0.00005	0.0001	0.0004	0.00006	0.00004	0.00006	1.41
Copper	mg/L	125	0	0.0002	0.0007	0.002	0.004	0.0008	0.0005	0.0007	1.65
Lead	mg/L	125	58	0.0001	0.0001	0.0005	0.003	0.00014	0.0003	0.00009	2.11
Molybdenum	mg/L	130	79	0.00005	0.00005	0.0002	0.0006	0.00009	0.0001	0.00007	1.71
Nickel	mg/L	125	0	0.0002	0.0005	0.001	0.002	0.0006	0.0003	0.0005	1.58
Selenium	mg/L	130	100	0.00005	0.00005	0.0001	0.0002	0.00006	0.00003	0.00006	1.40
Zinc	mg/L	125	7	0.0003	0.0029	0.009	0.011	0.004	0.003	0.003	2.47
Ammonia	mg/L	130	37	0.005	0.03	0.11	0.35	0.04	0.05	0.02	3.15
Chloride	mg/L	130	3	0.05	0.4	1.76	3.4	0.64	0.58	0.49	2.03
Sulphate	mg/L	130	1	0.10	0.6	1.56	6.9	0.78	0.70	0.66	1.67
TDS	mg/L	111	0	10.0	21	45.5	60.0	24.6	11.0	22.5	1.51

Table 3.2-3 Summary of Measured Baseline Water Quality

Water Concentrations - All Locations											
Constituent	**Units**	**N**	**N<MDL**	**Min**	**Median**	**95th**	**Max**	**Avg**	**St. Dev.**	**GM**	**GSD**
Calcium	mg/L	130	0	1.1	2.4	5.7	7.8	2.8	1.4	2.5	1.53
Note: N = number of samples. N<MDL = number of samples reported as less than the method detection limit. Min = minimum; 95th = 95th percentile; Max = maximum; Avg = average. St.Dev = standard deviation; GM = geometric mean; GSD = geometric standard deviation. µg/L = micrograms per litre; Bq/L = Bequerels per litre; mg/L – milligrams per litre											
Source:Appendix 5C, data from 2007, 2008, 2009; data reported as less than the method detection limit were treated as equal to ½ the method detection limit. Cadmium low level detection results from 2013 and provided in Attachement 5C-1											

The average values presented in Table 3.2-3 were used to represent the background concentrations for the site.

3.2.4 Baseline Sediment Quality

Sampling for sediment baseline quality was described in Appendix 5C for sampling completed in 2007, 2008, and 2009. Table 3.2-4 provides a summary of the data collected from the site for the COPC. Appendix 5C provides a summary of all of the measured data for sediment.

Table 3.2-4 Summary of Measured Baseline Sediment Quality

Sediment Concentrations - All Locations											
Constituent	**Units**	**N**	**N< MDL**	**Min**	**Median**	**95th**	**Max**	**Avg**	**St. Dev.**	**GM**	**GSD**
Uranium	µg/g dw	110	0	0.40	2.5	5.6	15.0	2.7	1.9	2.2	1.9
Thorium-230	Bq/g dw	110	15	0.01	0.04	0.09	0.16	0.04	0.02	0.04	1.9
Lead-210	Bq/g dw	110	34	0.01	0.06	0.19	0.38	0.07	0.07	0.05	2.6
Radium-226	Bq/g dw	110	5	0.005	0.04	0.08	0.10	0.04	0.02	0.03	1.8
Polonium-210	Bq/g dw	110	1	0.005	0.08	0.22	0.45	0.09	0.07	0.07	2.2
Arsenic	µg/g dw	110	0	1.2	7.4	15.0	45.0	8.0	6.4	6.2	2.1
Cadmium	µg/g dw	110	48	0.05	0.10	0.40	0.60	0.17	0.14	0.12	2.3
Cobalt	µg/g dw	110	0	1.2	5.4	10.6	30.0	5.6	3.7	4.7	1.9
Copper	µg/g dw	110	0	1.7	15.5	48.7	130.0	21.1	20.2	14.0	2.6
Lead	µg/g dw	110	0	2.4	11.0	17.0	20.0	10.2	4.9	8.7	1.8
Molybdenum	µg/g dw	110	3	0.05	0.60	3.1	17.0	1.3	2.2	0.7	2.8
Nickel	µg/g dw	110	0	4.7	22.0	40.6	71.0	22.2	12.1	18.6	1.9
Selenium	µg/g dw	110	22	0.05	0.30	0.90	1.3	0.36	0.29	0.24	2.7

Table 3.2-4 Summary of Measured Baseline Sediment Quality

Sediment Concentrations - All Locations											
Constituent	**Units**	**N**	**N< MDL**	**Min**	**Median**	**95th**	**Max**	**Avg**	**St. Dev.**	**GM**	**GSD**
Zinc	µg/g dw	110	0	13.0	63.5	130.0	440.0	69.9	59.7	54.9	2.0

Notes:
N = number of samples.
N<MDL = number of samples reported as less than the method detection limit.
Min = minimum; 95th = 95th percentile; Max = maximum; Avg = average.
St.Dev = standard deviation; GM = geometric mean; GSD = geometric standard deviation.
µg/g = micrograms per gram; Bq/g = Bequerels per gram; dw = dry weight basis

Source: Appendix 5C, data from 2007, 2008, 2009; data reported as less than the method detection limit were treated as equal to ½ the method detection limit

The average values presented in Table 3.2-4 were used to represent the background concentrations for the site.

3.2.5 Baseline Terrestrial Vegetation Quality

Baseline sampling was completed for terrestrial vegetation, as reported in Appendix 6B. Vegetation types sampled included: lichen, browse (willow, bog birch, and blueberry vegetation), forage (*Carex* sp. and sedge), and berry (blueberry and other berries). Tables 3.2-5 to 3.2-8 summarize the measured data for the baseline terrestrial vegetation at the site. Appendix 6B provides a summary of all of the measured data for terrestrial vegetation.

Table 3.2-5 Summary of Measured Baseline Terrestrial Vegetation Quality - Lichen

Lichen Concentrations - All Locations											
Constituent	**Units**	**N**	**N<MDL**	**Min**	**Median**	**95th**	**Max**	**Avg**	**St. Dev.**	**GM**	**GSD**
Uranium	µg/g ww	65	7	0.003	0.02	0.28	0.35	0.05	0.1	0.02	3.5
Thorium-230	Bq/g dw	43	35	0.00025	0.0025	0.006	0.006	0.002	0.00	0.002	3.0
Lead-210	Bq/g dw	43	0	0.26	0.47	0.62	0.71	0.47	0.10	0.46	1.3
Radium-226	Bq/g dw	43	2	0.0005	0.003	0.014	0.018	0.005	0.004	0.003	3.2
Polonium-210	Bq/g dw	43	0	0.15	0.40	0.64	0.65	0.42	0.10	0.41	1.3
Arsenic	µg/g ww	65	9	0.01	0.07	0.19	0.31	0.08	0.05	0.07	2.1
Cadmium	µg/g ww	65	0	0.02	0.10	0.22	0.30	0.11	0.06	0.10	1.9
Cobalt	µg/g ww	65	14	0.03	0.10	0.52	1.4	0.18	0.23	0.11	2.5
Copper	µg/g ww	65	0	0.56	1.7	4.5	6.2	1.9	1.1	1.7	1.6
Lead	µg/g ww	65	0	0.10	0.47	1.6	3.0	0.61	0.50	0.49	1.9
Molybdenum	µg/g ww	65	17	0.02	0.08	0.64	3.4	0.23	0.47	0.11	3.1

Table 3.2-5 Summary of Measured Baseline Terrestrial Vegetation Quality - Lichen

Lichen Concentrations - All Locations											
Constituent	**Units**	**N**	**N<MDL**	**Min**	**Median**	**95th**	**Max**	**Avg**	**St. Dev.**	**GM**	**GSD**
Nickel	µg/g ww	65	3	0.15	0.82	2.7	9.1	1.1	1.2	0.76	2.1
Selenium	µg/g ww	65	65	0.03	0.30	0.40	0.40	0.27	0.15	0.18	3.2
Zinc	µg/g ww	65	0	10.9	24.0	40.0	60.0	26.2	9.2	24.7	1.7

Notes:
N = number of samples.
N<MDL = number of samples reported as less than the method detection limit.
Min = minimum; 95th = 95th percentile; Max = maximum; Avg = average.
St.Dev = standard deviation; GM = geometric mean; GSD = geometric standard deviation.
µg/g = micrograms per gram; Bq/g = Bequerels per gram; ww = wet weight basis

Source: Appendix 6B, data from 2007, 2008, and 2009; data reported as less than the method detection limit was treated as equal to ½ the method detection limit. Sample Kig4-3 collected in 2008 was removed as an outlier from the database and was not included in the summary statistics.

Table 3.2-6 Summary of Measured Baseline Terrestrial Vegetation Quality - Browse

Browse Concentrations - All Locations											
Constituent	**Units**	**N**	**N<MDL**	**Min**	**Median**	**95th**	**Max**	**Avg**	**St. Dev.**	**GM**	**GSD**
Uranium	µg/g ww	18	17	0.005	0.005	0.006	0.010	0.005	0.001	0.005	1.2
Thorium-230	Bq/g dw	18	6	0.0002	0.0006	0.0012	0.002	0.0006	0.0004	0.0005	1.9
Lead-210	Bq/g dw	18	0	0.057	0.11	0.16	0.16	0.11	0.03	0.10	1.3
Radium-226	Bq/g dw	18	0	0.0012	0.004	0.008	0.014	0.004	0.003	0.004	1.8
Polonium-210	Bq/g dw	18	0	0.045	0.088	0.13	0.13	0.09	0.03	0.09	1.4
Arsenic	g/g ww	18	17	0.025	0.025	0.04	0.13	0.03	0.02	0.03	1.5
Cadmium	µg/g ww	18	0	0.08	0.58	4.2	5.0	1.2	1.5	0.62	3.5
Cobalt	µg/g ww	18	0	0.05	0.38	0.97	1.0	0.42	0.30	0.30	2.7
Copper	µg/g ww	18	0	4.3	6.5	10.0	12.0	6.8	2.1	6.5	1.4
Lead	µg/g ww	18	0	0.04	0.09	0.20	0.21	0.11	0.06	0.09	1.7
Molybdenum	µg/g ww	18	14	0.05	0.05	0.52	0.60	0.12	0.16	0.08	2.3
Nickel	µg/g ww	18	0	0.57	1.7	3.5	4.1	1.8	0.96	1.6	1.7
Selenium	µg/g ww	18	18	0.025	0.025	0.025	0.025	0.025	0	0.025	1.0
Zinc	µg/g ww	18	0	32	120	312	380	132	98	104	2.0

Notes:
N = number of samples.
N<MDL = number of samples reported as less than the method detection limit.
Min = minimum; 95th = 95th percentile; Max = maximum; Avg = average.
St.Dev = standard deviation; GM = geometric mean; GSD = geometric standard deviation.
µg/g = micrograms per gram; Bq/g = Bequerels per gram; ww = wet weight basis; dw = dry weight basis

Source: Appendix 6B, data from 2007, 2008, and 2009 for willow, bog birch, and blueberry vegetation; data reported as less than the method detection limit was treated as equal to ½ the method detection limit.

Table 3.2-7 Summary of Measured Baseline Terrestrial Vegetation Quality - Forage

Forage Concentrations - All Locations											
Constituent	**Units**	**N**	**N<MDL**	**Min**	**Median**	**95th**	**Max**	**Avg**	**St. Dev.**	**GM**	**GSD**
Uranium	µg/g ww	66	19	0.002	0.01	0.30	13.9	0.4	2.0	0.02	5.9
Thorium-230	Bq/g dw	43	31	0.00025	0.0025	0.010	0.12	0.006	0.02	0.002	3.2
Lead-210	Bq/g dw	43	0	0.007	0.20	0.42	0.46	0.22	0.12	0.16	2.5
Radium-226	Bq/g dw	43	1	0.0005	0.004	0.015	0.26	0.01	0.04	0.004	2.7
Polonium-210	Bq/g dw	43	0	0.012	0.15	0.31	0.38	0.17	0.10	0.13	2.3
Arsenic	µg/g ww	66	28	0.01	0.03	0.14	0.51	0.05	0.08	0.03	2.4
Cadmium	µg/g ww	66	2	0.005	0.04	0.12	0.18	0.05	0.04	0.04	2.1
Cobalt	µg/g ww	66	10	0.005	0.09	0.52	0.87	0.16	0.18	0.09	2.9
Copper	µg/g ww	66	0	0.37	2.0	3.9	4.4	2.2	0.94	2.0	1.6
Lead	µg/g ww	66	3	0.02	0.15	0.59	0.95	0.23	0.21	0.14	2.8
Molybdenum	µg/g ww	66	2	0.01	0.30	1.8	4.8	0.57	0.82	0.29	3.3
Nickel	µg/g ww	66	0	0.06	0.68	2.4	12.5	1.1	1.6	0.74	2.1
Selenium	µg/g ww	66	66	0.025	0.20	0.30	0.40	0.18	0.10	0.13	2.6
Zinc	µg/g ww	66	0	3.6	16.7	60	100	22.2	17.6	18.0	1.9

Notes:
N = number of samples.
N<MDL = number of samples reported as less than the method detection limit.
Min = minimum; 95th = 95th percentile; Max = maximum; Avg = average.
St.Dev = standard deviation; GM = geometric mean; GSD = geometric standard deviation.
µg/g = micrograms per gram; Bq/g = Bequerels per gram; ww = wet weight basis; dw = dry weight basis

Source: Appendix 6B, data from 2007, 2008, and 2009 for Carex sp. and sedge; data reported as less than the method detection limit was treated as equal to ½ the method detection limit.

Table 3.2-8 Summary of Measured Baseline Terrestrial Vegetation Quality - Berry

Berry Concentrations - All Locations											
Constituent	**Units**	**N**	**N<MDL**	**Min**	**Median**	**95th**	**Max**	**Avg**	**St. Dev.**	**GM**	**GSD**
Uranium	µg/g ww	51	48	0.001	0.001	0.005	0.006	0.002	0.002	0.001	1.9
Thorium-230	Bq/g dw	18	17	0.0002	0.002	0.0025	0.0025	0.002	0.001	0.001	2.5
Lead-210	Bq/g dw	18	2	0.001	0.01	0.04	0.09	0.01	0.02	0.01	3.0
Radium-226	Bq/g dw	18	5	0.00025	0.001	0.002	0.004	0.0012	0.0009	0.0009	2.0
Polonium-210	Bq/g dw	18	0	0.002	0.007	0.013	0.014	0.007	0.004	0.006	1.8
Arsenic	µg/g ww	51	50	0.005	0.005	0.025	0.026	0.009	0.008	0.007	1.9
Cadmium	µg/g ww	51	21	0.0025	0.01	0.19	0.36	0.04	0.08	0.01	4.3
Cobalt	µg/g ww	51	40	0.005	0.01	0.06	0.07	0.02	0.02	0.01	1.8
Copper	µg/g ww	51	0	0.54	0.70	4.7	7.0	1.3	1.5	0.92	2.0
Lead	µg/g ww	51	44	0.005	0.010	0.03	0.05	0.012	0.008	0.011	1.5
Molybdenum	µg/g ww	51	7	0.005	0.04	0.26	1.5	0.09	0.21	0.04	3.1
Nickel	µg/g ww	51	5	0.05	0.20	1.8	3.0	0.49	0.61	0.27	2.9
Selenium	µg/g ww	51	51	0.025	0.10	0.10	0.10	0.09	0.03	0.08	1.7
Zinc	µg/g ww	51	0	1.3	19.0	34	40.4	17.0	11.7	11.0	3.1

Notes:
N = number of samples.
N<MDL = number of samples reported as less than the method detection limit.
Min = minimum; 95th = 95th percentile; Max = maximum; Avg = average.
St.Dev = standard deviation; GM = geometric mean; GSD = geometric standard deviation.
µg/g = micrograms per gram; Bq/g = Bequerels per gram; ww = wet weight basis; dw = dry weight basis

Source: Appendix 6B, data from 2007, 2008, and 2009 for blueberry and other berries; data reported as less than the method detection limit was treated as equal to ½ the method detection limit.

3.2.6 Baseline Aquatic Vegetation Quality

Sampling for macrophyte (aquatic vegetation) baseline quality was described in Appendix 5C for sampling completed in 2009. Table 3.2-9 provides a summary of the data collected from the site for the COPC. Appendix 5C provides a summary of all of the measured data for aquatic vegetation.

Table 3.2-9 Summary of Measured Baseline Aquatic Vegetation Quality

Aquatic Vegetation Concentrations - All Locations											
Constituent	**Units**	**N**	**N<MDL**	**Min**	**Median**	**95th**	**Max**	**Avg**	**St. Dev.**	**GM**	**GSD**
Uranium	µg/g dw	30	11	0.005	0.045	0.50	0.86	0.13	0.20	0.04	6.2
Thorium-230	Bq/g dw	13	13	0.0005	0.0005	0.0015	0.003	0.0007	0.0007	0.0005	2.6
Lead-210	Bq/g dw	30	0	0.013	0.042	0.18	0.30	0.07	0.06	0.05	2.2
Radium-226	Bq/g dw	30	0	0.001	0.004	0.015	0.05	0.007	0.009	0.004	2.3
Polonium-210	Bq/g dw	30	0	0.012	0.032	0.13	0.16	0.05	0.04	0.04	2.2
Arsenic	µg/g ww	30	2	0.025	0.44	17.0	31.0	4.7	7.5	0.68	10.7
Cadmium	µg/g ww	30	2	0.005	0.055	0.46	0.71	0.13	0.17	0.06	4.0
Cobalt	µg/g ww	30	1	0.005	0.31	3.0	5.0	0.87	1.2	0.29	5.7
Copper	µg/g ww	30	0	2.2	4.5	16.0	27.0	6.1	5.3	4.9	1.8
Lead	µg/g ww	30	0	0.01	0.35	3.6	7.2	1.0	1.6	0.28	6.4
Molybdenum	µg/g ww	30	0	0.1	0.6	1.7	2.3	0.75	0.54	0.57	2.2
Nickel	µg/g ww	30	0	0.2	1.2	5.1	5.7	1.7	1.5	1.1	2.6
Selenium	µg/g ww	30	22	0.025	0.025	0.16	0.19	0.05	0.05	0.04	1.9
Zinc	µg/g ww	30	0	11.0	26.0	41.8	50.0	26.4	9.1	24.9	1.4

Notes:
N = number of samples.
N<MDL = number of samples reported as less than the method detection limit.
Min = minimum; 95th = 95th percentile; Max = maximum; Avg = average.
St.Dev = standard deviation; GM = geometric mean; GSD = geometric standard deviation.
µg/g = micrograms per gram; Bq/g = Bequerels per gram; dw = dry weight basis

Source: Appendix 5C, data from 2007, 2008, and 2009; data reported as less than the method detection limit was treated as equal to ½ the method detection limit. For shoots and roots combined.

3.2.7 Baseline Benthic Invertebrate Quality

Benthic invertebrates were collected from Judge Sissons Lake for analysis, as reported in Appendix 5C-1. In 2013, three soft-bodied benthic invertebrate samples and four hard-bodied benthic invertebrate samples were collected and analyzed. Species analyzed included stonefly nymph, mayfly nymph, caddisfly larva, crane fly larva, snails, aquatic worms, and diving beetle. Table 3.2-10 summarizes the measured data for the baseline benthic invertebrate quality at the site. Appendix 5C-1 provides a summary of all of the measured data for benthic invertebrates.

Table 3.2-10 Summary of Measured Baseline Benthic Invertebrate Quality

Benthic Invertebrate Concentrations - All Locations, Soft- and Hard-Bodied											
Constituent	**Units**	**N**	**N<MDL**	**Min**	**Median**	**95th**	**Max**	**Avg**	**St. Dev.**	**GM**	**GSD**
Uranium	µg/g ww	6	0	0.044	0.094	0.14	0.14	0.094	0.04	0.087	1.5
Thorium-230	Bq/g ww	3	3	0.003	0.003	0.004	0.004	0.003	0.0004	0.003	1.1
Lead-210	Bq/g ww	4	3	0.007	0.007	0.014	0.015	0.009	0.004	0.008	1.5
Radium-226	Bq/g ww	3	2	0.001	0.002	0.003	0.003	0.002	0.001	0.002	1.5
Polonium-210	Bq/g ww	4	0	0.025	0.033	0.08	0.09	0.045	0.03	0.039	1.7
Arsenic	µg/g ww	6	0	0.21	0.78	1.2	1.2	0.73	0.46	0.58	2.2
Cadmium	µg/g ww	6	0	0.04	0.11	0.31	0.32	0.15	0.13	0.10	2.8
Cobalt	µg/g ww	6	0	0.19	0.35	0.58	0.60	0.39	0.16	0.36	1.5
Copper	µg/g ww	6	0	2.7	10.7	23.9	24.0	12.4	10.7	7.8	3.1
Lead	µg/g ww	6	0	0.07	0.23	0.41	0.45	0.24	0.13	0.21	1.9
Molybdenum	µg/g ww	6	0	0.15	0.17	0.20	0.20	0.17	0.02	0.17	1.1
Nickel	µg/g ww	6	0	0.72	1.03	1.6	1.7	1.10	0.34	1.06	1.3
Selenium	µg/g ww	6	0	0.08	0.18	0.28	0.29	0.18	0.09	0.16	1.7
Zinc	µg/g ww	6	0	8.8	14.8	26.4	29.5	16.3	7.1	15.2	1.5

Notes:
N = number of samples.
N<MDL = number of samples reported as less than the method detection limit.
Min = minimum; 95th = 95th percentile; Max = maximum; Avg = average.
St.Dev = standard deviation; GM = geometric mean; GSD = geometric standard deviation.
µg/g = micrograms per gram; Bq/g = Bequerels per gram; ww = wet weight basis

Source: Appendix 5C-1, data from 2013; data reported as less than the method detection limit was treated as equal to ½ the method detection limit.

3.2.8 Baseline Insect Quality

Insects were collected from around the RAA and Mine LAA for analysis, as reported in Appendix 6C. In 2008, three samples were collected using Malaise traps, and in 2009, one sample was collected using a Skeeter-Vac. Flying insects, such as mosquitoes, midges, and blackflies, were captured, although the insect species were not specified. Table 3.2-11 summarizes the measured data for the baseline insect quality at the site. Appendix 6C provides a summary of all of the measured data for insects.

Table 3.2-11 Summary of Measured Baseline Insect Quality

Insect Concentrations - All Locations											
Constituent	**Units**	**N**	**N<MDL**	**Min**	**Median**	**95th**	**Max**	**Avg**	**St. Dev.**	**GM**	**GSD**
Uranium	µg/g ww	4	0	0.01	0.02	0.055	0.060	0.03	0.02	0.02	2.2
Thorium-230	Bq/g ww*	1	1	0.045	0.045	0.045	0.045	0.045	-	0.045	-
Lead-210	Bq/g ww*	1	1	0.011	0.011	0.011	0.011	0.011	-	0.011	-
Radium-226	Bq/g ww*	1	1	0.009	0.009	0.009	0.009	0.009	-	0.009	-
Polonium-210	Bq/g ww*	1	1	0.015	0.015	0.015	0.015	0.015	-	0.015	-
Arsenic	µg/g ww	4	1	0.025	0.033	0.069	0.075	0.042	0.023	0.038	1.6
Cadmium	µg/g ww	4	0	0.57	1.1	1.7	1.8	1.1	0.51	1.1	1.6
Cobalt	µg/g ww	4	1	0.03	0.08	0.12	0.13	0.08	0.04	0.07	1.8
Copper	µg/g ww	4	0	24.0	28.6	30.1	30.2	27.9	2.8	27.7	1.1
Lead	µg/g ww	4	0	0.15	0.37	0.78	0.83	0.43	0.30	0.35	2.1
Molybdenum	µg/g ww	4	0	0.59	0.73	0.97	1.00	0.76	0.19	0.74	1.3
Nickel	µg/g ww	4	0	0.32	0.53	2.2	2.5	0.97	1.0	0.69	2.4
Selenium	µg/g ww	4	0	0.52	0.75	0.79	0.80	0.70	0.12	0.69	1.2
Zinc	µg/g ww	4	0	109	151	618	700	278	282	204	2.3

Notes:
N = number of samples.
N<MDL = number of samples reported as less than the method detection limit.
Min = minimum; 95th = 95th percentile; Max = maximum; Avg = average.
St.Dev = standard deviation; GM = geometric mean; GSD = geometric standard deviation.
µg/g = micrograms per gram; Bq/g = Bequerels per gram; ww = wet weight basis
* = converted to a wet weight basis using an assumed moisture content of 70% (Gray and Bradley, 2005)

Source: Appendix 6C, data from 2008, and 2009; data reported as less than the method detection limit was treated as equal to ½ the method detection limit.

3.2.9 Baseline Fish Quality

Sampling for fish baseline quality was described in Appendix 5C for sampling completed in 2008 and 2009. A number of fish species were sampled, including Arctic grayling, cisco, lake trout, and white roundfish. Flesh and bone samples were analyzed separately, and only flesh data was considered for the assessment. However, all fish species were grouped together for the summary statistics presented in Table 3.2-12. Appendix 5C provides a summary of all of the measured data for fish.

Table 3.2-12 Summary of Measured Baseline Fish Quality

Fish Concentrations - All Locations											
Constituent	**Units**	**N**	**N<MDL**	**Min**	**Median**	**95th**	**Max**	**Avg**	**St. Dev.**	**GM**	**GSD**
Uranium	µg/g ww	56	32	0.0005	0.0005	0.03	0.16	0.007	0.02	0.001	4.6
Thorium-230	Bq/g ww	56	54	0.00004	0.0005	0.001	0.001	0.0004	0.0003	0.0002	3.3
Lead-210	Bq/g ww	56	50	0.0005	0.001	0.01	0.045	0.0027	0.008	0.0010	2.7
Radium-226	Bq/g ww	56	46	0.000025	0.00025	0.0005	0.002	0.0002	0.0003	0.0001	3.2
Polonium-210	Bq/g ww	56	28	0.0001	0.0003	0.001	0.002	0.0004	0.0004	0.0003	2.3
Arsenic	µg/g ww	56	6	0.005	0.02	0.05	0.05	0.03	0.01	0.02	1.9
Cadmium	µg/g ww	56	52	0.001	0.001	0.002	0.005	0.0012	0.0007	0.0011	1.4
Cobalt	µg/g ww	56	0	0.002	0.005	0.013	0.019	0.006	0.004	0.005	1.7
Copper	µg/g ww	56	0	0.12	0.26	0.58	0.80	0.32	0.14	0.29	1.5
Lead	µg/g ww	56	36	0.001	0.001	0.012	0.024	0.003	0.005	0.002	2.5
Molybdenum	µg/g ww	56	55	0.010	0.010	0.010	0.030	0.010	0.003	0.010	1.2
Nickel	µg/g ww	56	13	0.005	0.02	0.08	0.12	0.020	0.022	0.014	2.2
Selenium	µg/g ww	56	0	0.15	0.26	0.46	0.59	0.28	0.09	0.27	1.4
Zinc	µg/g ww	56	0	3.5	5.6	11.3	16.0	6.4	2.7	6.0	1.4

Notes:
N = number of samples.
N<MDL = number of samples reported as less than the method detection limit.
Min = minimum; 95th = 95th percentile; Max = maximum; Avg = average.
St.Dev = standard deviation; GM = geometric mean; GSD = geometric standard deviation.
µg/g = micrograms per gram; Bq/g = Bequerels per gram; ww = wet weight basis

Source: Appendix 5C, data from 2008, and 2009; data reported as less than the method detection limit was treated as equal to ½ the method detection limit; data for flesh only; all fish species grouped.

3.2.10 Baseline Bird Concentrations

Small mammals were captured for chemical tissue analysis in 2009, as reported in Appendix 6C. As part of that survey, several birds were inadvertently captured, and chemical analysis of the tissues were completed. The three Savannah sparrow specimens were collected from around the LAA (locations SIS-3 and SIS-5) and a reference location. Table Table 3.2-13 summarizes the measured data for sparrow at the site. Appendix 6C provides a summary of all of the measured data for sparrow.

Table 3.2-13 Summary of Measured Baseline Sparrow Quality

Sparrow Concentrations - All Locations											
Constituent	**Units**	**N**	**N<MDL**	**Min**	**Median**	**95th**	**Max**	**Avg**	**St. Dev.**	**GM**	**GSD**
Uranium	µg/g ww	3	3	0.001	0.001	0.001	0.001	0.001	0	0.001	1.0
Thorium-230	Bq/g ww	3	3	0.001	0.001	0.001	0.001	0.001	0	0.001	1.0
Lead-210	Bq/g ww	3	0	0.020	0.029	0.079	0.084	0.044	0.035	0.037	2.1
Radium-226	Bq/g ww	3	0	0.001	0.002	0.002	0.002	0.002	0.001	0.002	1.5
Polonium-210	Bq/g ww	3	0	0.006	0.014	0.024	0.025	0.015	0.010	0.013	2.0
Arsenic	µg/g ww	3	3	0.010	0.010	0.010	0.010	0.010	0	0.010	1.0
Cadmium	µg/g ww	3	0	0.005	0.015	0.018	0.018	0.013	0.007	0.011	2.0
Cobalt	µg/g ww	3	0	0.012	0.019	0.034	0.036	0.022	0.012	0.020	1.7
Copper	µg/g ww	3	0	2.2	2.5	2.9	2.9	2.5	0.35	2.5	1.1
Lead	µg/g ww	3	0	0.018	0.021	0.030	0.031	0.023	0.007	0.023	1.3
Molybdenum	µg/g ww	3	0	0.08	0.08	0.13	0.13	0.10	0.03	0.09	1.3
Nickel	µg/g ww	3	0	0.07	0.09	0.17	0.18	0.11	0.06	0.10	1.6
Selenium	µg/g ww	3	0	0.08	0.29	0.34	0.35	0.24	0.14	0.20	2.2
Zinc	µg/g ww	3	0	17.0	21.0	27.3	28.0	22.0	5.6	21.5	1.3

Notes:
N = number of samples.
N<MDL = number of samples reported as less than the method detection limit.
Min = minimum; 95th = 95th percentile; Max = maximum; Avg = average.
St.Dev = standard deviation; GM = geometric mean; GSD = geometric standard deviation.
µg/g = micrograms per gram; Bq/g = Bequerels per gram; ww = wet weight basis

Source: Appendix 6C, data from 2007, 2008, and 2009; data reported as less than the method detection limit was treated as equal to ½ the method detection limit.

3.2.11 Baseline Animal Concentrations

Small mammals were captured for chemical tissue analysis in 2009, as reported in Appendix 6C. There were 16 lemming specimens collected from around the LAA (Kiggavik and Sissons sites). Table 3.2-14 summarizes the measured data for lemming at the site. Appendix 6C provides a summary of all of the measured data for these small mammals.

Table 3.2-14 Summary of Measured Baseline Lemming Quality

Lemming Concentrations - All Locations											
Constituent	**Units**	**N**	**N<MDL**	**Min**	**Median**	**95th**	**Max**	**Avg**	**St. Dev.**	**GM**	**GSD**
Uranium	µg/g ww	16	9	0.001	0.001	0.19	0.73	0.05	0.2	0.003	5.8
Thorium-230	Bq/g ww	16	15	0.001	0.001	0.003	0.006	0.001	0.001	0.001	1.6
Lead-210	Bq/g ww	16	0	0.005	0.066	0.14	0.15	0.07	0.05	0.05	2.9
Radium-226	Bq/g ww	16	8	0.0005	0.00075	0.15	0.58	0.037	0.14	0.001	5.8
Polonium-210	Bq/g ww	16	0	0.002	0.0045	0.030	0.036	0.010	0.010	0.006	2.4
Arsenic	µg/g ww	16	10	0.010	0.010	0.06	0.11	0.02	0.03	0.02	2.0
Cadmium	µg/g ww	16	0	0.005	0.048	0.13	0.17	0.06	0.04	0.04	2.4
Cobalt	µg/g ww	16	0	0.028	0.084	0.17	0.18	0.10	0.05	0.08	1.7
Copper	µg/g ww	16	0	1.6	2.3	2.6	2.7	2.3	0.30	2.3	1.1
Lead	µg/g ww	16	0	0.015	0.065	0.18	0.22	0.07	0.06	0.05	2.4
Molybdenum	µg/g ww	16	3	0.025	0.095	0.90	1.4	0.27	0.39	0.12	3.5
Nickel	µg/g ww	16	0	0.07	0.19	0.33	0.37	0.20	0.08	0.18	1.6
Selenium	µg/g ww	16	0	0.03	0.04	0.12	0.27	0.06	0.06	0.05	1.7
Zinc	µg/g ww	16	0	16.0	26.5	34.8	37.0	27.7	5.5	27.1	1.2

Notes:
N = number of samples.
N<MDL = number of samples reported as less than the method detection limit.
Min = minimum; 95th = 95th percentile; Max = maximum; Avg = average.
St.Dev = standard deviation; GM = geometric mean; GSD = geometric standard deviation.
µg/g = micrograms per gram; Bq/g = Bequerels per gram; ww = wet weight basis

Source: Appendix 6C, data from 2007, 2008, and 2009; data reported as less than the method detection limit was treated as equal to ½ the method detection limit.

There were 11 vole specimens collected from the site, as described in Appendix 6C. Table 3.2-15 summarizes the measured data for vole at the site. Appendix 6C provides a summary of the vole data as well.

Table 3.2-15 Summary of Measured Baseline Vole Quality

Vole Concentrations - All Locations											
Constituent	**Units**	**N**	**N<MDL**	**Min**	**Median**	**95th**	**Max**	**Avg**	**St. Dev.**	**GM**	**GSD**
Uranium	µg/g ww	11	6	0.001	0.001	0.23	0.43	0.04	0.13	0.003	6.7
Thorium-230	Bq/g ww	11	9	0.001	0.001	0.004	0.004	0.001	0.001	0.001	1.7
Lead-210	Bq/g ww	11	1	0.002	0.057	0.22	0.29	0.09	0.08	0.06	3.6
Radium-226	Bq/g ww	11	2	0.0005	0.001	0.006	0.008	0.002	0.002	0.001	2.3
Polonium-210	Bq/g ww	11	0	0.001	0.013	0.032	0.037	0.015	0.011	0.010	3.0
Arsenic	µg/g ww	11	9	0.01	0.01	0.02	0.02	0.012	0.004	0.011	1.3
Cadmium	µg/g ww	11	1	0.0025	0.01	0.015	0.017	0.010	0.004	0.009	1.6
Cobalt	µg/g ww	11	0	0.036	0.063	0.087	0.094	0.060	0.019	0.058	1.4
Copper	µg/g ww	11	0	2.0	2.4	3.8	5.0	2.6	0.83	2.5	1.3
Lead	µg/g ww	11	0	0.02	0.03	0.10	0.10	0.05	0.03	0.04	2.0
Molybdenum	µg/g ww	11	0	0.12	0.13	0.21	0.22	0.15	0.03	0.14	1.2
Nickel	µg/g ww	11	0	0.07	0.13	0.20	0.22	0.14	0.04	0.13	1.4
Selenium	µg/g ww	11	0	0.05	0.14	0.23	0.24	0.13	0.07	0.12	1.7
Zinc	µg/g ww	11	0	24.0	28.0	34.5	35.0	28.9	3.6	28.7	1.1

Notes:
N = number of samples.
N<MDL = number of samples reported as less than the method detection limit.
Min = minimum; 95th = 95th percentile; Max = maximum; Avg = average.
St.Dev = standard deviation; GM = geometric mean; GSD = geometric standard deviation.
µg/g = micrograms per gram; Bq/g = Bequerels per gram; ww = wet weight basis

Source: Appendix 6C, data reported as less than the method detection limit was treated as equal to ½ the method detection limit.

Appendix 6C describes the collection of 76 caribou tissue samples (bone, kidney, liver, and muscle) from 27 separate caribou provided by Baker Lake hunters in February and May 2009. Samples were collected from animals that had either been stored or recently killed for personal use. Table 3.2-16 summarizes the measured data for caribou collected from the Baker Lake area. Attachment B of this report provides a summary of the caribou data, as well as a comparison with caribou collected from other areas in Northern Canada.

Table 3.2-16 Summary of Measured Baseline Caribou Quality

Caribou Muscle Concentrations - All Locations											
Constituent	**Units**	**N**	**N<MDL**	**Min**	**Median**	**95th**	**Max**	**Avg**	**St. Dev.**	**GM**	**GSD**
Uranium	µg/g ww	23	8	0.0005	0.002	0.031	0.17	0.013	0.04	0.003	5.2
Thorium-230	Bq/g ww	23	22	0.00004	0.00005	0.0001	0.0005	0.0001	0.0001	0.0001	1.7
Lead-210	Bq/g ww	23	14	0.0005	0.0005	0.004	0.007	0.001	0.002	0.001	2.3
Radium-226	Bq/g ww	23	6	0.000025	0.0001	0.00020	0.00035	0.00011	0.00008	0.00009	2.1
Polonium-210	Bq/g ww	23	0	0.004	0.026	0.039	0.046	0.027	0.008	0.025	1.6
Arsenic	µg/g ww	23	1	0.005	0.040	0.050	0.060	0.035	0.014	0.031	1.8
Cadmium	µg/g ww	23	2	0.001	0.006	0.02	0.07	0.010	0.014	0.007	2.5
Cobalt	µg/g ww	23	0	0.002	0.006	0.02	0.02	0.008	0.005	0.006	1.9
Copper	µg/g ww	23	0	0.74	2.1	4.0	4.2	2.4	0.81	2.2	1.4
Lead	µg/g ww	23	0	0.002	0.014	0.038	0.049	0.016	0.012	0.012	2.5
Molybdenum	µg/g ww	23	13	0.01	0.01	0.07	0.46	0.05	0.09	0.02	2.8
Nickel	µg/g ww	23	7	0.005	0.020	0.088	0.160	0.032	0.036	0.019	2.9
Selenium	µg/g ww	23	0	0.10	0.18	0.21	0.22	0.17	0.02	0.17	1.2
Zinc	µg/g ww	23	0	18.0	41.0	61.8	62.0	40.4	14.2	37.7	1.5

Notes:
N = number of samples.
N<MDL = number of samples reported as less than the method detection limit.
Min = minimum; 95th = 95th percentile; Max = maximum; Avg = average.
St.Dev = standard deviation; GM = geometric mean; GSD = geometric standard deviation.
µg/g = micrograms per gram; Bq/g = Bequerels per gram; dw = dry weight basis; ww = wet weight basis

Source: Appendix 6C, data from 2009; data reported as less than the method detection limit was treated as equal to ½ the method detection limit.

Tissue, bone, and kidney samples were analyzed for a muskox harvested in May 2011, as reported in Appendix 6C. Table 3.2-17 summarizes the measured data for muskox from the general study area. Appendix 6C provides a summary of all of the measured data for muskox.

Table 3.2-17 Summary of Measured Baseline Muskox Quality

Muskox Tissue Concentrations				
Constituent	**Units**	**N**	**N<MDL**	**Value**
Uranium	µg/g ww	1	0	0.03
Thorium-230	Bq/g ww	1	1	<0.0001
Lead-210	Bq/g ww	1	0	0.002
Radium-226	Bq/g ww	1	1	<0.00005
Polonium-210	Bq/g ww	1	0	0.0087
Arsenic	µg/g ww	1	0	0.02
Cadmium	µg/g ww	1	1	<0.002
Cobalt	µg/g ww	1	0	0.003
Copper	µg/g ww	1	0	1.2
Lead	µg/g ww	1	0	0.099
Molybdenum	µg/g ww	1	1	<0.02
Nickel	µg/g ww	1	0	0.01
Selenium	µg/g ww	1	0	0.11
Zinc	µg/g ww	1	0	40
Notes: N = number of samples. N<MDL = number of samples reported as less than the method detection limit. µg/g = micrograms per gram; Bq/g = Bequerels per gram; dw = dry weight basis; ww = wet weight basis				
Source: Appendix 6C.				

3.3 Bioavailability Factors

Sediment samples in the Kiggavik study area were collected at various lake locations in the Pointer Lake, Shack Lake, Judge Sissons Lake, and Squiggly Lake watersheds. Tessier sequential extraction tests (Tessier et al., 1979; Davidson et al., 1994) were performed on 21 composite samples. The sediment samples were analyzed following the six-step Tessier extraction procedure to measure metal and radionuclide binding in the sediments. The test procedure differentiates between: 1) water soluble: fraction dissolved in distilled water; 2) ion exchangeable: fraction that is weakly adsorbed and released by ion-exchange processes; 3) fraction bound to carbonates and released by organic acid treatment; 4) fraction bound to Fe and Mn oxides: (also called the “reducible fraction”); 5) sulphide and/or organic matter bound fraction (also called the “oxidizable fraction”); and 6) residual fraction: not extractable by the above methods.

Normally, the first four fractions are considered available for sediment/water column exchange. The specific constituents that are contained in these fractions are expected to be reversibly bound and are in equilibrium with concentrations in the porewater phase. The extractable fraction of the COPC

metals ranges from near 0% (antimony, silver, thorium) to near 100% (cadmium), but most constituents are less than 50% extractable. As expected, the extractable fraction of the oxy-anions (arsenate, molybdate, selenate) is primarily absorbed onto iron and manganese oxides. On the average, 78% of the extractable arsenic, 61% of extractable molybdenum, and 58% of the extractable selenium was partitioned in the Fe/Mn oxide fraction. Cations (cobalt, copper, nickel, zinc) are also mainly associated with the Fe/Mn oxide fraction, but the distribution includes significant speciation as ion exchangeable species and carbonates. Similar results were obtained by Sutherland (2002). Of the COPC metals, uranium is an exception inasmuch as most (69%) of the exchangeable uranium is associated with carbonates, while the rest (31%) is adsorbed onto iron/manganese oxides. Amongst the radionuclides, Pb-210 and Po-210 are mainly associated with iron oxides while Ra-226 speciation includes a significant ion exchangeable fraction.

To analyze for relationships between variables, the extraction data were subjected to K-mean cluster analysis. The technique involves grouping data of similar characteristics (Romesburg, 2004). K-mean cluster analysis is widely recognized as a powerful hierarchical algorithm that can identify similarities in large data sets. This chemometric technique involves grouping data of similar chemical or physico-chemical characteristics. It has replaced simple statistical functions such as the arithmetic mean in data mining applications. In the present investigation, the total metal concentrations (µg/g) in the sediment and the extractable fractions of 21 observations were employed. The environmentally mobile fraction was defined as the sum of the first four extraction measurements (water soluble, ion exchangeable, carbonate bound and adsorbed to Fe/Mn oxides) in each sequential Tessier analysis. The logarithmic values of observations were used to avoid bias between the variable types (concentration vs % extraction). The Euclidean distance between the extraction data sets was chosen as the cluster measure. This is one of the most common means of measuring similarities between data. It is computed as follows in equation (3.3-1):

$$D(X_i, X_j) = \left[\sum_{l=1}^{N} (X_{i,l} - X_{j,l})^2 \right]^{1/2} \qquad (3.3\text{-}1)$$

where:
$D(X_i,X_j)$ = distance between the i-th and j-th set of observations
$X_{i,l}$ = the l-th observation in the j-th set
l = 1,2,3….N

The value of N was 2 (total metal content and % extractable content) in the present case.

A K-mean cluster analysis was completed for the Judge Sissons Lake samples. The main reason for selecting the K-mean cluster approach was the small sediment sample size in Judge Sissons Lake and the relatively high frequency of apparent outliers either in the sediment concentration or the % extraction data. The mean distance between uncorrelated locations varied depending on the constituent. Using a one-sided significance test, the critical distance at the 95% confidence level was estimated. On this basis, the dataset naturally formed distinct clusters. Cluster size depended on the

particular constituent in question. The clustering results indicate that the sediment composition and extractability in the Kiggavik area are similar and the data from several locations may be combined with sediments from Judge Sissons Lake to derive characteristic composition and extractability information.

The extractable fractions together with the total metal concentrations were used, in turn, to establish solid phase "background" concentrations and linear sorption coefficients (Kd values) for further analysis of water and sediment quality impact of discharges on Judge Sissons Lake. The sorption coefficient (K_d), also known as the partition or distribution coefficient, is an estimate of the partitioning of the constituents between the solid (i.e., sediment) and liquid (porewater) phases. The results of the k-mean cluster analysis to estimate extractable fractions are summarized in Table 3.3-1.

Table 3.3-1 Summary of Calculated Site-Specific Extractable Fraction

COPC	Extractable Fraction
Uranium	0.460
Thorium-230	1.0[a]
Lead-210	0.626[b]
Radium-226	0.334
Polonium-210	0.050[b]
Arsenic	0.174
Cadmium	1.0[a]
Cobalt	0.309
Copper	0.166
Lead	0.235
Molybdenum	0.150
Nickel	0.157
Selenium	0.243
Zinc	0.459

Note:
a assumed
b adjusted to 0.4 for extractable fraction during calibration

The site-specific extractable fractions were used in the sediment model and for the bioavailability of ingested soil and sediment.

3.4 Emissions

3.4.1 Atmospheric Emissions

Emissions for the atmospheric dispersion model are discussed in detail (Appendix 4B Air Quality); however, a brief overview is provided here. During the construction, operational years and into the post-closure phase of the Project, sources of atmospheric emissions may include:

- drilling and blasting;
- ore and waste rock handling;
- maintenance activities (i.e., grading and bulldozing);
- wind erosion of stockpiled ore and mine waste rock;
- non-road mining equipment;
- vehicular transportation;
- milling operations; and
- supporting activities such as power generation.

Activities like material handling along with heavy equipment operation are also primary emission sources during construction and quarrying. Additional activities such as aggregate crushing and screening are also emission sources during these phases.

Various phases of development were assessed over the lifetime of the Project including construction, mining stages, decommissioning and post-decommissioning. Long-term air quality effects were assessed for four (4) production scenarios associated with the scheduled activities of the Project over its lifetime. These various scenarios assess the development of the three open pit mines (East Zone, Centre Zone and Main Zone) at the Kiggavik site and both the Andrew Lake open pit mine and End Grid underground mine at the Sissons site. Production schedules for mining and milling were both considered in deriving the modelling scenarios to consider a range of operating conditions. A description of each of the modelled scenarios is provided in Appendix 4B Air Quality.

3.4.2 Liquid Effluent

The liquid effluent associated with the Kiggavik Project is characterized by quality and flow profiles. The three aspects defining the liquid effluent assumptions throughout the assessment period are the discharge scenario, the operation scenario, and the closure scenario. The following sections provide a screening process to ascertain a conservative assessment scenario that covers the potential options outlined for discharge scenarios, operation scenarios, and closure scenarios. The scenario selected will be considered for the ERA and HHRA and the results will encompass the range of results from the various other potential options for discharge, operation, and closure.

3.4.2.1 Discharge Scenarios

Three discharge scenarios were considered for the water treatment plants (WTPs) associated with the Kiggavik Project, as described below. Refer to Figure 2.7-1 for the segments of Judge Sissons Lake.

- Option 1: Kiggavik WTP discharges to segment JSL-2 and Sissons WTP discharges to segment JSL-8
- Option 2: Kiggavik WTP and Sissons WTP discharge to segment JSL-2
- Option 3: the Kiggavik WTP is expanded to also treat effluent from Sissons, and the expanded Kiggavik WTP discharges to segment JSL-2

Table 3.4-1 summarizes the maximum predicted COPC water concentrations in each segment of Judge Sissons Lake for the three options. For comparison purposes, the maximum concentration of each COPC in each segment from the three scenarios has been identified. Since there is a direct discharge from the Sissons WTP to JSL-8 in Option 1, the maximum concentrations of the three scenarios generally occurs in this area of the lake. There is less constituent load into JSL-2 for this option, compared with Options 2 and 3. Options 2 and 3 clearly have a greater influence on the water concentrations in the vicinity of the JSL-2, since these two options exclusively discharge effluent to this location. This influence extends to the outlet of Judge Sissons Lake (JSL-4), where Options 2 and 3 also have the highest predicted maximum monthly mean water concentrations for the constituents.

Table 3.4-1 Maximum of Monthly Mean Water Concentrations in JSL Segments – Option 1 (O1) vs Option 2 (O2) vs Option 3 (O3)

COPC	Units	JSL-1			JSL-2			JSL-3		
		O1	O2	O3	O1	O2	O3	O1	O2	O3
Uranium	µg/L	0.3	0.4	**0.5**	0.2	0.4	**0.5**	0.2	0.3	**0.4**
Thorium-230	Bq/L	0.013	**0.014**	**0.014**	0.009	**0.010**	**0.010**	0.009	0.009	0.009
Lead-210	Bq/L	0.024	**0.025**	**0.025**	0.016	**0.018**	0.017	0.014	**0.015**	**0.015**
Radium-226	Bq/L	0.009	0.009	0.009	0.006	0.006	0.006	0.005	0.005	0.005
Polonium-210	Bq/L	0.011	0.011	0.011	0.007	**0.008**	**0.008**	0.006	**0.007**	**0.007**
Arsenic	µg/L	0.6	**0.7**	0.6	0.5	**0.6**	**0.6**	0.4	**0.5**	0.4
Cadmium	µg/L	0.1	0.1	0.1	0.2	0.2	0.2	0.2	0.2	0.2
Cobalt	µg/L	0.2	0.2	0.2	0.1	0.1	**0.2**	0.1	0.1	0.1
Copper	µg/L	1.8	1.8	1.8	1.1	1.1	1.1	1.0	1.0	1.0
Lead	µg/L	0.2	0.2	**0.3**	0.2	0.2	0.2	0.1	0.1	**0.2**
Molybdenum	µg/L	2.9	**3.2**	2.9	3.6	**4.0**	3.6	2.9	**3.1**	2.8
Nickel	µg/L	1.4	1.4	1.4	1.0	1.0	1.0	0.8	0.8	**0.9**
Selenium	µg/L	0.3	0.3	0.3	0.3	0.3	0.3	0.2	**0.3**	0.2
Zinc	µg/L	8.8	8.8	8.8	5.5	5.5	5.5	4.9	4.9	4.9
Ammonia (Unionized)	mg/L	0.003	0.003	0.003	0.006	0.006	0.006	0.007	**0.008**	**0.008**
Chloride	mg/L	19.6	24.7	**26.0**	51.4	55.6	**55.9**	64.6	67.0	**68.7**
Sulphate	mg/L	72	73	**79**	167	166	**183**	199	199	**223**
TDS	mg/L	151	163	**193**	299	309	**382**	357	365	**460**
Hardness (as $CaCO_3$)	mg/L	55	**63**	61	121	**125**	122	147	**150**	148
COPC	**Units**	**JSL-4**			**JSL-5**			**JSL-6**		
		O1	**O2**	**O3**	**O1**	**O2**	**O3**	**O1**	**O2**	**O3**
Uranium	µg/L	0.2	0.2	**0.3**	0.2	0.2	**0.3**	0.2	0.2	0.2
Thorium-230	Bq/L	0.008	**0.009**	**0.009**	0.009	0.009	0.009	0.011	0.011	0.011
Lead-210	Bq/L	0.014	**0.015**	**0.015**	0.016	0.016	0.016	0.020	0.020	0.020
Radium-226	Bq/L	0.005	0.005	0.005	0.006	0.006	0.006	0.007	0.007	0.007
Polonium-210	Bq/L	0.006	**0.007**	**0.007**	0.007	0.007	0.007	0.009	0.009	0.009
Arsenic	µg/L	0.4	0.4	0.4	0.4	0.4	0.4	0.4	0.4	0.4
Cadmium	µg/L	0.1	0.1	**0.2**	0.1	0.1	0.1	0.1	0.1	0.1
Cobalt	µg/L	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1
Copper	µg/L	1.0	1.0	1.0	1.1	1.1	1.1	1.4	1.4	1.4
Lead	µg/L	0.1	0.1	0.1	0.1	0.1	**0.2**	0.2	0.2	0.2

Table 3.4-2 Maximum of Monthly Mean Water Concentrations in JSL Segments – Option 1 (O1) vs Option 2 (O2) vs Option 3 (O3)

COPC	Units	JSL-4			JSL-5			JSL-6		
		O1	O2	O3	O1	O2	O3	O1	O2	O3
Molybdenum	µg/L	2.5	**2.6**	2.3	**2.1**	2.0	1.9	**1.2**	1.1	1.0
Nickel	µg/L	0.8	0.8	0.8	0.9	0.9	0.9	1.1	1.1	1.1
Selenium	µg/L	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2
Zinc	µg/L	5.0	5.0	5.0	5.6	5.6	5.6	7.2	7.2	7.2
Ammonia (Unionized)	mg/L	0.005	0.005	0.005	0.004	0.004	0.004	0.0010	0.0011	0.0011
Chloride	mg/L	58.1	59.4	**60.9**	46.4	46.1	**47.0**	**8.9**	8.0	8.2
Sulphate	mg/L	177	178	**198**	137	137	**153**	22.2	22.3	**24.2**
TDS	mg/L	322	328	**411**	260	264	**328**	72.3	75.1	**83.7**
Hardness (as $CaCO_3$)	mg/L	132	**135**	132	105	**107**	105	25.0	**26.2**	25.6
COPC	**Units**	**JSL-7**			**JSL-8**					
		O1	**O2**	**O3**	**O1**	**O2**	**O3**			
Uranium	µg/L	**1.4**	1.0	1.0	**0.8**	0.3	0.3			
Thorium-230	Bq/L	**0.060**	0.059	0.059	**0.018**	0.017	0.017			
Lead-210	Bq/L	**0.109**	0.107	0.107	**0.032**	0.030	0.030			
Radium-226	Bq/L	**0.040**	0.039	0.039	**0.012**	0.011	0.011			
Polonium-210	Bq/L	0.049	0.049	0.049	0.014	0.014	0.014			
Arsenic	µg/L	**2.1**	2.0	2.0	**0.7**	0.6	0.6			
Cadmium	µg/L	0.1	0.1	0.1	0.1	0.1	0.1			
Cobalt	µg/L	0.6	0.6	0.6	0.2	0.2	0.2			
Copper	µg/L	7.8	7.8	7.8	2.2	2.2	2.2			
Lead	µg/L	1.0	1.0	1.0	0.3	0.3	0.3			
Molybdenum	µg/L	**3.7**	2.4	2.3	**2.4**	1.2	1.1			
Nickel	µg/L	5.9	5.9	5.9	1.7	1.7	1.7			
Selenium	µg/L	1.0	1.0	1.0	0.3	0.3	0.3			
Zinc	µg/L	39.0	39.0	39.0	11.0	11.0	11.0			
Ammonia (Unionized)	mg/L	0.002	0.002	0.002	**0.0017**	0.0016	0.0016			
Chloride	mg/L	**47.2**	25.2	26.2	**33.8**	15.1	15.8			
Sulphate	mg/L	67.7	65.0	**70.3**	45.0	42.8	**46.3**			
TDS	mg/L	335	329	**353**	140	131	**148**			
Hardness (as $CaCO_3$)	mg/L	**114.4**	106.9	105.4	**54.9**	46.5	45.5			

Notes:
on 1
on 2
on 3
Bold underlining indicates the maximum value for the three options.

Table 3.4-3 summarizes the maximum predicted sediment concentrations in each segment of Judge Sissons Lake for the three options. Similar to the results for water, Options 2 and 3 have a greater influence on the sediment concentrations in the vicinity of the JSL-2, since these two options exclusively discharge effluent to this location. For sediment, differences in the maximum predicted concentrations between the three options are generally minor.

Table 3.4-3 Maximum of Monthly Mean Sediment Concentrations in JSL Segments – Option 1 (O1) vs Option 2 (O2) vs Option 3 (O3)

COPC	Units	JSL-1			JSL-2			JSL-3		
		O1	O2	O3	O1	O2	O3	O1	O2	O3
Uranium	µg/g	2.7	2.8	**3.2**	2.7	3.4	**4.1**	2.7	3.1	**3.5**
Thorium-230	Bq/g	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04
Lead-210	Bq/g	0.07	0.07	0.07	0.07	0.07	0.07	0.07	0.07	0.07
Radium-226	Bq/g	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04
Polonium-210	Bq/g	0.07	0.07	0.07	0.07	0.07	0.07	0.07	0.07	0.07
Arsenic	µg/g	8.3	**8.5**	**8.5**	8.8	**9.1**	**9.1**	8.6	**8.8**	**8.8**
Cadmium	µg/g	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2
Cobalt	µg/g	5.6	5.6	5.6	6.5	6.5	**6.7**	6.4	6.4	**6.5**
Copper	µg/g	21.9	**22.0**	**22.0**	21.9	22.0	**22.1**	21.5	21.6	**21.7**
Lead	µg/g	9.4	9.4	9.4	9.6	9.6	**9.7**	9.4	9.4	**9.6**
Molybdenum	µg/g	3.7	**4.1**	3.8	5.2	**5.9**	5.4	4.4	**4.8**	4.3
Nickel	µg/g	22.4	22.4	22.4	**22.8**	22.7	**22.8**	**22.5**	22.4	22.4
Selenium	µg/g	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5
Zinc	µg/g	**64.1**	63.6	63.6	**67.4**	66.9	66.9	**64.7**	64.1	64.1
COPC	**Units**	**JSL-4**			**JSL-5**			**JSL-6**		
		O1	**O2**	**O3**	**O1**	**O2**	**O3**	**O1**	**O2**	**O3**
Uranium	µg/g	2.7	3.0	**3.3**	2.8	2.8	**3.0**	2.7	2.7	2.7
Thorium-230	Bq/g	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04
Lead-210	Bq/g	0.07	0.07	0.07	0.07	0.07	0.07	0.07	0.07	0.07
Radium-226	Bq/g	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04
Polonium-210	Bq/g	0.07	0.07	0.07	0.07	0.07	0.07	0.07	0.07	0.07
Arsenic	µg/g	8.5	**8.7**	**8.7**	8.4	8.5	**8.6**	8.2	8.2	8.2
Cadmium	µg/g	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2
Cobalt	µg/g	6.3	6.3	**6.4**	6.2	6.2	**6.3**	5.6	5.6	5.6
Copper	µg/g	21.7	**21.9**	**21.9**	22.0	22.1	**22.2**	21.7	**21.8**	**21.8**
Lead	µg/g	9.5	9.5	**9.6**	9.6	9.6	9.6	9.4	9.4	9.4
Molybdenum	µg/g	3.8	**4.1**	3.7	**3.3**	**3.3**	3.1	**2.3**	**2.3**	2.2
Nickel	µg/g	**22.6**	22.5	22.5	22.7	22.7	22.7	22.4	22.4	22.4
Selenium	µg/g	0.5	0.5	0.5	0.4	**0.5**	0.4	0.4	0.4	0.4

Table 3.4-3 Maximum of Monthly Mean Sediment Concentrations in JSL Segments – Option 1 (O1) vs Option 2 (O2) vs Option 3 (O3)

COPC	Units	JSL-1			JSL-2			JSL-3		
		O1	O2	O3	O1	O2	O3	O1	O2	O3
Zinc	µg/g	**66.9**	66.3	66.3	**69.2**	68.5	68.5	**63.3**	63.2	63.2
COPC	**Units**	**JSL-7**			**JSL-8**					
		O1	**O2**	**O3**	**O1**	**O2**	**O3**			
Uranium	µg/g	**2.9**	2.7	2.7	**3.5**	2.7	2.7			
Thorium-230	Bq/g	0.05	0.05	0.05	0.04	0.04	0.04			
Lead-210	Bq/g	0.07	0.07	0.07	0.07	0.07	0.07			
Radium-226	Bq/g	0.04	0.04	0.04	0.04	0.04	0.04			
Polonium-210	Bq/g	0.07	0.07	0.07	0.07	0.07	0.07			
Arsenic	µg/g	8.5	**8.6**	**8.6**	**8.3**	8.2	8.2			
Cadmium	µg/g	0.2	0.2	0.2	0.2	0.2	0.2			
Cobalt	µg/g	5.6	5.6	5.6	5.6	5.6	5.6			
Copper	µg/g	23.7	**23.8**	**23.8**	22.1	**22.2**	**22.2**			
Lead	µg/g	9.4	9.4	9.4	9.4	9.4	9.4			
Molybdenum	µg/g	**2.3**	1.9	1.8	**3.0**	2.0	1.9			
Nickel	µg/g	**23.8**	23.7	23.7	22.5	22.5	22.5			
Selenium	µg/g	0.5	0.5	0.5	0.4	0.4	0.4			
Zinc	µg/g	**76.1**	75.4	75.4	**65.9**	65.4	65.4			

Notes:
on 1
on 2
on 3
Bold underlining indicates the maximum value for the three options.

Discharge scenario Option 1 is selected for evaluation in the ecological and human health risk assessment. The predicted water and sediment quality for the three discharge scenarios is similar, and Option 1 was selected because it has two separate discharge locations and the potential to impact two areas of the lake. The risks associated with this option are considered to be representative or conservative of any of the options that may be selected.

3.4.2.2 Operation Scenarios

Two operation scenarios were considered for the assessment:

- Project Scenario: 13 years of operation for the Kiggavik Project (operation ends June 2026)

- Extended Scenario: 25 years of operation for the Kiggavik Project (operation ends June 2038)

Table 3.4-4 summarizes the maximum predicted water concentrations in each segment of Judge Sissons Lake for the two operation scenarios. A comparison of the values in each segment shows that in most cases, the maximum predicted water concentrations are the same for the two operation scenarios. However, when the values are different, the maximum predicted water concentrations are higher for the Extended Scenario.

Table 3.4-4 Maximum of Monthly Mean Water Concentrations in JSL Segments – Project Scenario vs Extended Scenario

COPC	Units	JSL-1		JSL-2		JSL-3		JSL-4	
		Pr	Ex	Pr	Ex	Pr	Ex	Pr	Ex
Uranium	µg/L	0.3	0.3	0.2	0.2	0.2	0.2	0.2	0.2
Thorium-230	Bq/L	0.013	0.013	0.009	0.009	0.008	0.008	0.008	0.008
Lead-210	Bq/L	0.024	0.024	0.016	0.016	0.014	0.014	0.014	0.014
Radium-226	Bq/L	0.009	0.009	0.006	0.006	0.005	0.005	0.005	0.005
Polonium-210	Bq/L	0.011	0.011	0.007	0.007	0.006	0.006	0.006	0.006
Arsenic	µg/L	0.6	0.6	0.5	0.5	0.4	0.4	0.4	0.4
Cadmium	µg/L	0.1	0.1	0.2	0.2	0.2	0.2	0.1	0.1
Cobalt	µg/L	0.2	0.2	0.1	0.1	0.1	0.1	0.1	0.1
Copper	µg/L	1.8	1.8	1.1	1.1	1.0	1.0	1.0	1.0
Lead	µg/L	0.2	0.2	0.2	0.2	0.1	0.1	0.1	0.1
Molybdenum	µg/L	2.7	**2.8**	3.3	**3.6**	2.7	**2.9**	2.3	**2.5**
Nickel	µg/L	1.4	1.4	1.0	1.0	0.8	0.8	0.8	0.8
Selenium	µg/L	0.3	0.3	0.3	0.3	0.2	**0.3**	0.2	0.2
Zinc	µg/L	8.8	8.8	5.5	5.5	4.9	4.9	5.0	5.0
Ammonia (Unionized)	mg/L	0.003	0.003	0.006	0.006	0.008	0.008	0.005	0.005

Table 3.4-4 Maximum of Monthly Mean Water Concentrations in JSL Segments – Project Scenario vs Extended Scenario

COPC	Units	JSL-1		JSL-2		JSL-3		JSL-4	
		Pr	Ex	Pr	Ex	Pr	Ex	Pr	Ex
Chloride	mg/L	19.9	19.9	51.3	**51.9**	65.2	**65.7**	58.4	**59.0**
Sulphate	mg/L	72	72	166	166	199	199	177	177
TDS	mg/L	148	**149**	299	299	355	**359**	322	**323**
Hardness (as $CaCO_3$)	mg/L	56	56	121	121	147	**148**	133	133
COPC	**Units**	**JSL-5**		**JSL-6**		**JSL-7**		**JSL-8**	
		Pr	**Ex**	**Pr**	**Ex**	**Pr**	**Ex**	**Pr**	**Ex**
Uranium	µg/L	0.2	0.2	0.2	0.2	1.4	1.4	0.8	0.8
Thorium-230	Bq/L	0.009	0.009	0.011	0.011	0.060	0.060	0.018	0.018
Lead-210	Bq/L	0.016	0.016	0.020	0.020	0.109	0.109	0.032	0.032
Radium-226	Bq/L	0.006	0.006	0.007	0.007	0.040	0.040	0.012	0.012
Polonium-210	Bq/L	0.007	0.007	0.009	0.009	0.049	0.049	0.014	0.014
Arsenic	µg/L	0.4	0.4	0.4	0.4	2.1	2.1	0.7	0.7
Cadmium	µg/L	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1
Cobalt	µg/L	0.1	0.1	0.1	0.1	0.6	0.6	0.2	0.2
Copper	µg/L	1.1	1.1	1.4	1.4	7.8	7.8	2.2	2.2
Lead	µg/L	0.1	0.1	0.2	0.2	1.0	1.0	0.3	0.3
Molybdenum	µg/L	1.9	**2.0**	1.1	**1.2**	3.6	**3.7**	2.4	2.4
Nickel	µg/L	0.9	0.9	1.1	1.1	5.9	5.9	1.7	1.7

COPC	Units	JSL-5		JSL-6		JSL-7		JSL-8	
		Pr	Ex	Pr	Ex	Pr	Ex	Pr	Ex
Selenium	µg/L	0.2	0.2	0.2	0.2	1.0	1.0	0.3	0.3
Zinc	µg/L	5.6	5.6	7.2	7.2	39.0	39.0	11.0	11.0
Ammonia (Unionized)	mg/L	0.004	0.004	0.0010	0.0010	0.002	0.002	0.0017	0.0017
Chloride	mg/L	46.5	**47.2**	9.0	**9.1**	47.0	**48.2**	33.6	**34.5**
Sulphate	mg/L	137	137	22.3	22.3	67.8	67.8	45.0	**45.2**
TDS	mg/L	260	**261**	71.6	**72.1**	332	**334**	138	**140**
Hardness (as $CaCO_3$)	mg/L	106	106	25.1	**25.3**	114.0	**115.2**	54.6	**55.4**

Note: results are presented for O1Pr14 and O1Ex14.

Pr – Project Scenario

Ex – Extended Scenario

Bold underlining indicates the maximum value for the two scenarios

Table 3.4-5 summarizes the maximum predicted sediment concentrations in each segment of Judge Sissons Lake for the two operation scenarios. A comparison of the values in each segment shows that in most cases, the maximum predicted sediment concentrations are the same for the two operation scenarios. However, when the values are different, the maximum predicted sediment concentrations are higher for the Extended Scenario, with the exception of copper, which has slightly higher maximum monthly mean predicted sediment concentrations in the Project Scenario.

Table 3.4-5 Maximum of Monthly Mean Sediment Concentrations in JSL Segments – Project Scenario vs Extended Scenario

COPC	Units	JSL-1		JSL-2		JSL-3		JSL-4	
		Pr	Ex	Pr	Ex	Pr	Ex	Pr	Ex
Uranium	µg/g	2.7	2.7	2.7	2.7	2.7	2.7	2.7	2.7
Thorium-230	Bq/g	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04
Lead-210	Bq/g	0.07	0.07	0.07	0.07	0.07	0.07	0.07	0.07
Radium-226	Bq/g	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04
Polonium-210	Bq/g	0.07	0.07	0.07	0.07	0.07	0.07	0.07	0.07
Arsenic	µg/g	8.3	8.3	8.6	**8.8**	8.5	**8.6**	8.4	**8.5**
Cadmium	µg/g	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2
Cobalt	µg/g	5.6	5.6	6.3	**6.5**	6.2	**6.4**	6.2	**6.3**
Copper	µg/g	**22.0**	21.9	**22.0**	21.9	**21.6**	21.5	**21.9**	21.7
Lead	µg/g	9.4	9.4	9.5	**9.6**	9.4	9.4	9.5	9.5
Molybdenum	µg/g	3.5	**3.7**	4.6	**5.2**	4.1	**4.4**	3.6	**3.8**
Nickel	µg/g	22.4	22.4	22.5	**22.8**	22.4	**22.5**	22.4	**22.6**
Selenium	µg/g	0.5	0.5	0.5	0.5	0.5	0.5	0.4	**0.5**
Zinc	µg/g	63.7	**64.1**	66.9	**67.4**	64.1	**64.7**	66.3	**66.9**

COPC	Units	JSL-5		JSL-6		JSL-7		JSL-8	
		Pr	Ex	Pr	Ex	Pr	Ex	Pr	Ex
Uranium	µg/g	2.7	**2.8**	2.7	2.7	2.8	**2.9**	3.2	**3.5**
Thorium-230	Bq/g	0.04	0.04	0.04	0.04	0.05	0.05	0.04	0.04
Lead-210	Bq/g	0.07	0.07	0.07	0.07	0.07	0.07	0.07	0.07
Radium-226	Bq/g	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04
Polonium-210	Bq/g	0.07	0.07	0.07	0.07	0.07	0.07	0.07	0.07
Arsenic	µg/g	8.3	**8.4**	8.2	8.2	**8.6**	8.5	8.2	**8.3**
Cadmium	µg/g	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2
Cobalt	µg/g	6.2	6.2	5.6	5.6	5.6	5.6	5.6	5.6
Copper	µg/g	**22.1**	22.0	**21.8**	21.7	**23.8**	23.7	**22.2**	22.1
Lead	µg/g	9.5	**9.6**	9.4	9.4	9.4	9.4	9.4	9.4

COPC	Units	JSL-5		JSL-6		JSL-7		JSL-8	
		Pr	Ex	Pr	Ex	Pr	Ex	Pr	Ex
Molybdenum	µg/g	3.1	**3.3**	2.3	2.3	2.2	**2.3**	2.9	**3.0**
Nickel	µg/g	22.6	**22.7**	22.4	22.4	23.7	**23.8**	22.4	**22.5**
Selenium	µg/g	0.4	0.4	0.4	0.4	0.5	0.5	0.4	0.4
Zinc	µg/g	68.5	**69.2**	63.2	**63.3**	75.4	**76.1**	65.4	**65.9**
Note: results are presented for O1Pr14 and O1Ex14. Pr – Project Scenario Ex – Extended Scenario **Bold underlining** indicates the maximum value for the two scenarios.									

Therefore, the Extended Scenario is selected for the operation component of the liquid effluent characterization and is evaluated in the ecological and human health risk assessment. The Extended Scenario represents the conservative operation scenario.

3.4.2.3 Closure Scenarios

Multiple closure scenarios were evaluated to address uncertainty associated with resources (and resulting tailings mass) and tailings consolidation rate. Therefore, four closure scenarios for the Kiggavik effluent were considered for the assessment:

- Base Case: for the Project Scenario only, East Zone and Centre Zone consolidation and decommissioning completed during the operational period, Main Zone TMF partially full at the end of the operational period

  - 14-years for Main Zone Consolidation (Kiggavik closure complete in 30 years)

  - 22-years for Main Zone Consolidation (Kiggavik closure complete in 37 years)

- Bounding Case: for the Extended Scenario only, the three TMFs are filled with tailings and no consolidation has occurred prior to the decommissioning period

  - 14-years for Main Zone Consolidation (Kiggavik closure complete in 42 years)

  - 22-years for Main Zone Consolidation (Kiggavik closure complete in 49 years)

Because the Extended Scenario was selected as the conservative operation scenario for the evaluation, only the two bounding cases for the Extended Scenario are considered further. Table 3.4-5 summarizes the maximum predicted water concentrations in each segment of Judge Sissons Lake for the two bounding case closure scenarios. The maximum predicted water concentration occurs during the operation period and therefore, the bounding scenarios do not affect the maximum water concentrations experienced by the project.

Table 3.4-6 Maximum of Monthly Mean Water Concentrations in JSL Segments – 14-Year Bounding Case Scenario vs 22-Year Bounding Case Scenario

COPC	Units	JSL-1		JSL-2		JSL-3		JSL-4	
		14-yr	22-yr	14-yr	22-yr	14-yr	22-yr	14-yr	22-yr
Uranium	µg/L	0.3	0.3	0.2	0.2	0.2	0.2	0.2	0.2
Thorium-230	Bq/L	0.013	0.013	0.009	0.009	0.008	0.009	0.008	0.008
Lead-210	Bq/L	0.024	0.024	0.016	0.016	0.014	0.014	0.014	0.014
Radium-226	Bq/L	0.009	0.009	0.006	0.006	0.005	0.005	0.005	0.005
Polonium-210	Bq/L	0.011	0.011	0.007	0.007	0.006	0.006	0.006	0.006
Arsenic	µg/L	0.6	0.6	0.5	0.5	0.4	0.4	0.4	0.4
Cadmium	µg/L	0.1	0.1	0.2	0.2	0.2	0.2	0.1	0.1
Cobalt	µg/L	0.2	0.2	0.1	0.1	0.1	0.1	0.1	0.1
Copper	µg/L	1.8	1.8	1.1	1.1	1.0	1.0	1.0	1.0
Lead	µg/L	0.2	0.2	0.2	0.2	0.1	0.1	0.1	0.1
Molybdenum	µg/L	2.8	2.9	3.6	3.6	2.9	2.9	2.5	2.5
Nickel	µg/L	1.4	1.4	1.0	1.0	0.8	0.8	0.8	0.8
Selenium	µg/L	0.3	0.3	0.3	0.3	0.3	0.2	0.2	0.2
Zinc	µg/L	8.8	8.8	5.5	5.5	4.9	4.9	5.0	5.0
Ammonia (Unionized)	mg/L	0.003	0.003	0.006	0.006	0.008	0.007	0.005	0.005
Chloride	mg/L	19.9	19.6	51.9	51.4	65.7	64.6	59.0	58.1
Sulphate	mg/L	72	72	166	167	199	199	177	177
TDS	mg/L	149	151	299	299	359	357	323	322
Hardness (as $CaCO_3$)	mg/L	56	55	121	121	148	147	133	132

COPC	Units	JSL-5		JSL-6		JSL-7		JSL-8	
		14-yr	22-yr	14-yr	22-yr	14-yr	22-yr	14-yr	22-yr
Uranium	µg/L	0.2	0.2	0.2	0.2	1.4	1.4	0.8	0.8
Thorium-230	Bq/L	0.009	0.009	0.011	0.011	0.060	0.060	0.018	0.018
Lead-210	Bq/L	0.016	0.016	0.020	0.020	0.109	0.109	0.032	0.032
Radium-226	Bq/L	0.006	0.006	0.007	0.007	0.040	0.040	0.012	0.012

COPC	Units	JSL-5		JSL-6		JSL-7		JSL-8	
		14-yr	22-yr	14-yr	22-yr	14-yr	22-yr	14-yr	22-yr
Polonium-210	Bq/L	0.007	0.007	0.009	0.009	0.049	0.049	0.014	0.014
Arsenic	µg/L	0.4	0.4	0.4	0.4	2.1	2.1	0.7	0.7
Cadmium	µg/L	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1
Cobalt	µg/L	0.1	0.1	0.1	0.1	0.6	0.6	0.2	0.2
Copper	µg/L	1.1	1.1	1.4	1.4	7.8	7.8	2.2	2.2
Lead	µg/L	0.1	0.1	0.2	0.2	1.0	1.0	0.3	0.3
Molybdenum	µg/L	2.0	2.1	1.2	1.2	3.7	3.7	2.4	2.4
Nickel	µg/L	0.9	0.9	1.1	1.1	5.9	5.9	1.7	1.7
Selenium	µg/L	0.2	0.2	0.2	0.2	1.0	1.0	0.3	0.3
Zinc	µg/L	5.6	5.6	7.2	7.2	39.0	39.0	11.0	11.0
Ammonia (Unionized)	mg/L	0.004	0.004	0.0010	0.0010	0.002	0.002	0.0017	0.0017
Chloride	mg/L	47.2	46.4	9.1	8.9	48.2	47.2	34.5	33.8
Sulphate	mg/L	137	137	22.3	22.2	67.8	67.7	45.2	45.0
TDS	mg/L	261	260	72.1	72.3	334	335	140	140
Hardness (as $CaCO_3$)	mg/L	106	105	25.3	25.0	115.2	114.4	55.4	54.9

Note: results are presented for O1Ex14 and O1Ex22.
14-yr – Bounding Case 14-years
22-yr – Bounding Case 22-years
Bold underlining indicates the maximum value for the two scenarios

Table 3.4-7 summarizes the maximum predicted sediment concentrations in each segment of Judge Sissons Lake for the two closure scenarios. Similar to water, the maximum predicted sediment concentration occurs during the operation period and therefore, the maximum values are the same for the two closure scenarios.

Although the maximum predicted concentrations for the two Kiggavik closure scenarios are equal, the Bounding Case 22-year consolidation results in effluent loads to the receiving environment for a longer amount of time. Therefore, the 22-year consolidation for the Bounding Case is selected for the closure component of the liquid effluent characterization and is evaluated in the ecological and human health risk assessment.

Table 3.4-7 Maximum of Monthly Mean Sediment Concentrations in JSL Segments – 14-Year Bounding Case Scenario vs 22-Year Bounding Scenario

COPC	Units	JSL-1		JSL-2		JSL-3		JSL-4	
		14-yr	22-yr	14-yr	22-yr	14-yr	22-yr	14-yr	22-yr
Uranium	µg/g	2.7	2.7	2.7	2.7	2.7	2.7	2.7	2.7
Thorium-230	Bq/g	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04
Lead-210	Bq/g	0.07	0.07	0.07	0.07	0.07	0.07	0.07	0.07
Radium-226	Bq/g	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04
Polonium-210	Bq/g	0.07	0.07	0.07	0.07	0.07	0.07	0.07	0.07
Arsenic	µg/g	8.3	8.3	8.8	8.8	8.6	8.6	8.5	8.5
Cadmium	µg/g	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2
Cobalt	µg/g	5.6	5.6	6.5	6.5	6.4	6.4	6.3	6.3
Copper	µg/g	21.9	21.9	21.9	21.9	21.5	21.5	21.7	21.7
Lead	µg/g	9.4	9.4	9.6	9.6	9.4	9.4	9.5	9.5
Molybdenum	µg/g	3.7	3.7	5.2	5.2	4.4	4.4	3.8	3.8
Nickel	µg/g	22.4	22.4	22.8	22.8	22.5	22.5	22.6	22.6
Selenium	µg/g	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5
Zinc	µg/g	64.1	64.1	67.4	67.4	64.7	64.7	66.9	66.9
COPC	**Units**	**JSL-5**		**JSL-6**		**JSL-7**		**JSL-8**	
		14-yr	**22-yr**	**14-yr**	**22-yr**	**14-yr**	**22-yr**	**14-yr**	**22-yr**
Uranium	µg/g	2.8	2.8	2.7	2.7	2.9	2.9	3.5	3.5
Thorium-230	Bq/g	0.04	0.04	0.04	0.04	0.05	0.05	0.04	0.04
Lead-210	Bq/g	0.07	0.07	0.07	0.07	0.07	0.07	0.07	0.07
Radium-226	Bq/g	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04
Polonium-210	Bq/g	0.07	0.07	0.07	0.07	0.07	0.07	0.07	0.07
Arsenic	µg/g	8.4	8.4	8.2	8.2	8.5	8.5	8.3	8.3
Cadmium	µg/g	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2
Cobalt	µg/g	6.2	6.2	5.6	5.6	5.6	5.6	5.6	5.6
Copper	µg/g	22.0	22.0	21.7	21.7	23.7	23.7	22.1	22.1
Lead	µg/g	9.6	9.6	9.4	9.4	9.4	9.4	9.4	9.4
Molybdenum	µg/g	3.3	3.3	2.3	2.3	2.3	2.3	3.0	3.0
Nickel	µg/g	22.7	22.7	22.4	22.4	23.8	23.8	22.5	22.5
Selenium	µg/g	0.4	0.4	0.4	0.4	0.5	0.5	0.4	0.4
Zinc	µg/g	69.2	69.2	63.3	63.3	76.1	76.1	65.9	65.9

Note: results are presented for O1Ex14 and O1Ex22.
14-yr – Bounding Case 14-years
22-yr – Bounding Case 22-years
Bold underlining indicates the maximum value for the two scenarios.

One closure scenario was assessed for the Sissons effluent, with decommissioning continuing for 7 years (Sissons closure complete June 2031).

3.4.2.4 Summary of Scenario Selected for Evaluation

Option 1 Extended Operation with 22-year consolidation (O1Ex22) was selected as the scenario for the evaluation of ecological and human health risks. Option 1 was selected due to several factors, though primarily because it has discharge to two separate locations and therefore, the risks associated with this option are considered to be representative or conservative of any of the options that may be selected. Option 1 considers the discharge of Kiggavik WTP to segment JSL-2 and discharge of Sissons WTP to segment JSL-8 (Figure 2.7-1). This scenario represents the most probable plan with respect to discharge locations. The previous sections illustrated that differences in COPC concentrations between the scenarios are sufficiently small that they are within the range of model accuracy.

Tables 3.4-7 through 3.4-9 summarize the water quality distributions assumed to characterize the Kiggavik WTP, Sissons WTP, and Kiggavik Reverse Osmosis (RO) effluent, respectively. Predicted WTP and RO effluent quality values were provided by AREVA. In order to develop reasonable probability distributions for the effluent sources, operational data from similar systems at the JEB, McClean, and Key Lake sites were evaluated and the observed variance in these datasets were used to define the variance for the Kiggavik Project effluents.

Figures 3.4-1 through 3.4-3 present the assumed monthly flows for the O1Ex22 scenario. The predicted monthly flows were provided by AREVA and these values were assumed to be the mode of a triangular distribution, with the minimum and maximum represented by +/- 33%. Assumptions for the other liquid effluent discharge scenarios are provided in Attachment C.

Table 3.4-8 Effluent Concentration Distributions for Kiggavik WTP Discharge

COPC	Lognormal Distribution Descriptors (mg/L or Bq/L)			
	Geometric Mean (GM)[a]	Geometric Standard Deviation (GSD)[b]	Minimum (-3 GSD)	Maximum (+3 GSD)
Uranium	0.002	2.24	0.0002	0.02
Thorium-230	0.011	2.05	0.0013	0.10
Lead-210	0.052	1.54	0.014	0.19
Radium-226	0.008	2.53	0.0005	0.13
Polonium-210	0.007	2.10	0.0007	0.06
Arsenic	0.021	1.76	0.004	0.11
Cadmium	0.007	1.44	0.0023	0.02
Cobalt	0.007	2.12	0.0007	0.07
Copper	0.002	2.41	0.0001	0.02
Lead	0.002	2.63	0.0001	0.04
Molybdenum	0.2	1.75	0.038	1.1
Nickel	0.02	1.57	0.005	0.08
Selenium	0.01	1.28	0.0047	0.02
Zinc	0.003	2.06	0.0003	0.03
Ammonia	17.3	1.36	6.9	44
Calcium	470	1.29	219	1010
Chloride	237	1.49	71	792
Sulphate	2199	1.29	1027	4708
TDS[c]	3115	1.31	1373	7068

Note: a - GM from predicted Kiggavik WTP data, if not specified. Kiggavik WTP values provided by AREVA were assumed to be geometric mean values for the distributions.

b – GSDs preferentially selected from Key Lake data (4/2009-9/2010) for uranium, radium-226, arsenic, copper, molybdenum, nickel, selenium, and ammonia. GSDs from McClean (2011) for the JEB WTP (future quality) were used in the absence of other data for thorium-230, lead-210, polonium-210, cadmium, cobalt, lead, zinc, chloride, sulphate, and TDS.

c – TDS calculated as the sum of the anions and cations (as available).

Table 3.4-9 Effluent Concentration Distributions for Sissons WTP Discharge

COPC	Lognormal Distribution Descriptors (mg/L or Bq/L)			
	Geometric Mean (GM)[a]	Geometric Standard Deviation (GSD)[b]	Minimum (-3 GSD)	Maximum (+3 GSD)
Uranium	0.034	2.24	0.003	0.38
Thorium-230	0.19	2.05	0.022	1.64
Lead-210	0.59	1.54	0.16	2.17
Radium-226	0.10	2.53	0.006	1.62
Polonium-210	0.17	2.10	0.018	1.58
Arsenic	0.018	1.76	0.003	0.10
Cadmium	0.0001	1.44	0.00004	0.0004
Cobalt	0.0003	2.12	0.00003	0.003
Copper	0.001	2.41	0.00007	0.014
Lead	0.0005	2.63	0.00003	0.009
Molybdenum	0.085	1.75	0.016	0.46
Nickel	0.001	1.57	0.0003	0.005
Selenium	0.004	1.28	0.002	0.008
Zinc	0.014	2.06	0.002	0.12
Ammonia	3.1	1.36	1.24	7.86
Calcium	336	1.29	157	721
Chloride	846.6	1.49	254	2825
Sulphate	167.0	1.29	78	357
TDS[c]	1528	1.31	673	3466

Note: a - GM from predicted Sissons WTP data, if not specified. Sisson WTP values provided by AREVA were assumed to be geometric mean values for the distributions.

b – GSDs preferentially selected from Key Lake data (4/2009-9/2010) for uranium, radium-226, arsenic, copper, molybdenum, nickel, selenium, and ammonia. GSDs from McClean (2011) for the JEB WTP (future quality) were used in the absence of other data for thorium-230, lead-210, polonium-210, cadmium, cobalt, lead, zinc, chloride, sulphate, and TDS.

c – TDS calculated as the sum of the anions and cations (as available).

Table 3.4-10 Effluent Concentration Distributions for Kiggavik RO Discharge

COPC	Lognormal Distribution Descriptors (mg/L or Bq/L)			
	Geometric Mean (GM)[a]	Geometric Standard Deviation (GSD)[b]	Minimum (-3 GSD)	Maximum (+3 GSD)
Uranium	0.0003	2.25	0.00003	0.003
Thorium-230	0.002	2.05	0.0002	0.02
Lead-210	0.005	1.54	0.001	0.02
Radium-226	0.013	2.42	0.0009	0.19
Polonium-210	0.002	2.10	0.0002	0.02
Arsenic	0.001	1.60	0.00025	0.00
Cadmium	0.0003	1.44	0.0001	0.001
Cobalt	0.0003	2.12	0.0000	0.003
Copper	0.0004	2.04	0.00005	0.003
Lead	0.0002	2.63	0.00001	0.004
Molybdenum	0.03	1.69	0.006	0.13
Nickel	0.001	2.31	0.0001	0.01
Selenium	0.001	2.42	0.000071	0.0142
Zinc	0.0001	2.28	0.00001	0.001
Ammonia	1.11	1.63	0.254	4.84
Calcium	1.06	1.29	0.494	2.28
Chloride	1.0	1.49	0.300	3.34
Sulphate	5.2	1.29	2.4	11.2
TDS[c]	11	1.31	4.7	24.0

Note: a - Kiggavik RO values provided by AREVA were assumed to be geometric mean values for the distributions. The ammonia GM value from the Midwest RO. The use of Midwest RO for ammonia GM is conservative.

b – GSDs from McClean (2011) for the JEB WTP (future quality) for all but nickel . The GSD for nickel was based on the Midwest RO GSD due to high variability in the JEB WTP. The Midwest RO GSD was based on the measured JEB dewatering well system discharge.

c – TDS calculated as the sum of the anions and cations (as available).

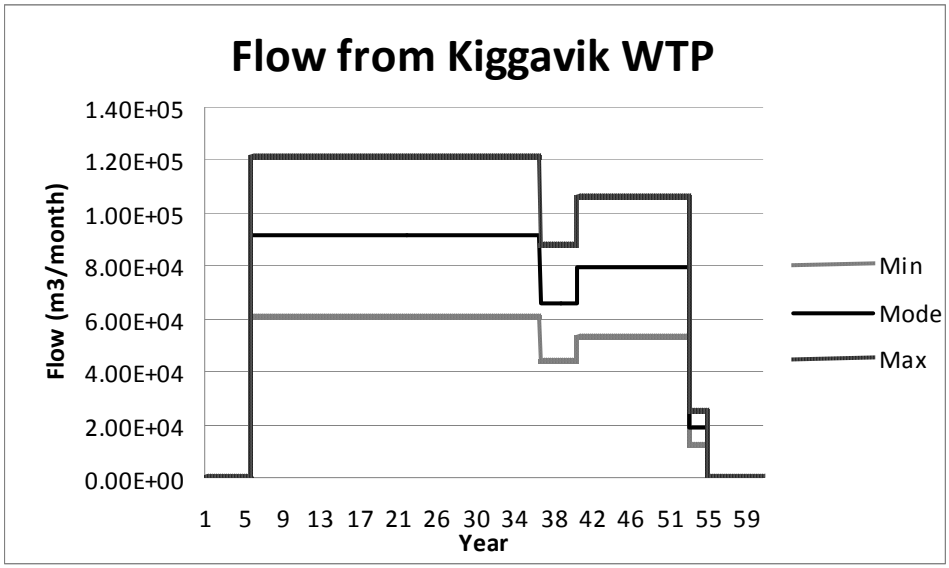


Figure 3.4-1 Assumed Flow Distributions for the Kiggavik WTP

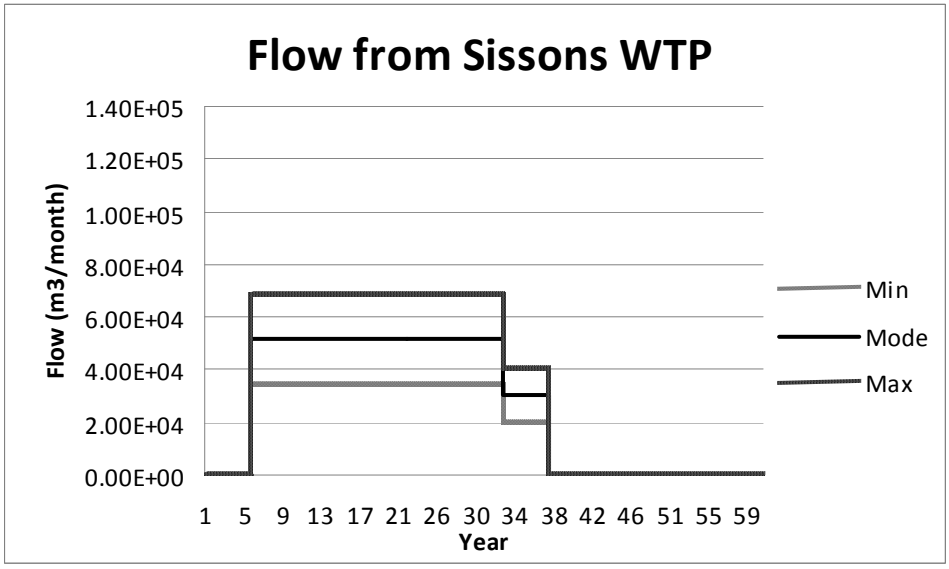


Figure 3.4-2 Assumed Flow Distributions for the Sissons WTP

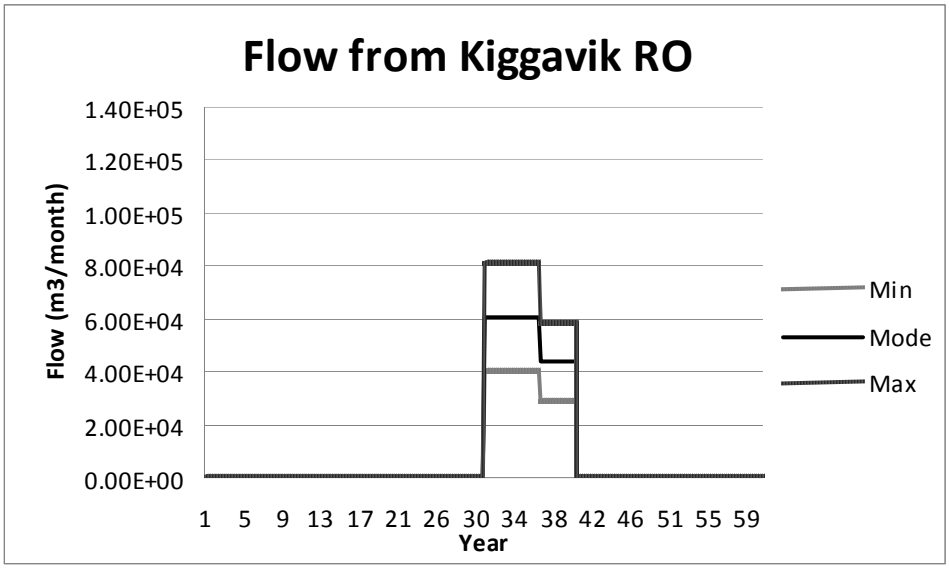


Figure 3.4-3 Assumed Flow Distributions for the Kiggavik RO

4 Receptor Characterization

One of the key considerations defining the scope of a risk assessment is the selection and characterization of ecological and human receptors. Receptors are used in risk assessments as endpoints for the assessment of the potential effects of a project or activity. Receptors are selected to represent the VECs and VSECs identified. In selecting receptors, it is important to identify plants, animals, and people that are likely to be most exposed to the effects of the Project, as well as those that may be important for other ecological or social reasons. This section details the ecological (aquatic and terrestrial) species and people selected for the assessment, and it provides the rationale behind their selection.

4.1 Ecological Receptors

The first step in the assessment of ecological receptors is to determine the ecological receptors to be examined. In an ecological risk assessment (ERA), it is not necessary to evaluate potential effects on all species that are present in the study area; rather, ecological receptors are generally chosen to capture various levels of exposure via the different types of diets that they consume. Ecological receptors are also selected if they are considered important: i) in the functioning of the ecosystem; ii) in the production of food for subsistence; or iii) due to their cultural or medicinal significance. As the Kiggavik Project will feature emissions to both the aquatic and atmospheric environments, several aquatic and terrestrial based species were selected to capture exposures via aquatic, atmospheric, and terrestrial pathways.

4.1.1 Aquatic Receptors

Aquatic receptors selected for this assessment, summarized below, cover all food chain levels in lake systems.

Phytoplankton
Zooplankton
Aquatic plants (pondweed)
Benthic invertebrates
Forage fish (burbot and white roundfish)
Predatory fish (Arctic grayling, cisco, and lake trout)

Aquatic receptors were evaluated in each segment of Judge Sissons Lake. The following sections provide the rationale for the selection.

4.1.1.1 Primary Producers

Primary producers are organisms that occupy the lowest level in the food chain. These organisms are generally plants which use solar energy and inorganic substances for growth and reproduction.

Aquatic Plants

Aquatic plants in most lake ecosystems (for example, pond weed) can constitute an important primary producer biomass. Aquatic plants are consumed by some species of aquatic birds and are therefore a component of the foodchain. Besides being an important food resource, aquatic plants provide habitat to aquatic organisms. For the radiological assessment, the leaves and roots of aquatic plants are considered to evaluate the dose from water and sediment, respectively.

Phytoplankton

Phytoplankton can also comprise an important component of the aquatic biomass in aquatic systems and are an important food sources to both aquatic and terrestrial species. Phytoplankton historically collected from the area lakes were rich in species typical of shallow tundra pond habitats. They included green algae, *Cyanobacteria, Cyanophytes, Chlrophytes* and *Chrysophytes* and diatoms (Urangesellschaft 1992).

Members of the division *Chlorophyta* have been studied extensively and are relatively common in most aquatic ecosystems in the north. Even though the overall contribution of *Chlorophyta* to aquatic ecosystems in the north is relatively small, *Chlorophyta* are a primary food resource for grazing zooplankton. *Chlorophyta* are one of the phytoplankton groups that are considered in this assessment; in the absence of data on *Chlorophyta,* other representative phytoplankton species are used. Therefore, the assessment considers phytoplankton as the overall ecological receptor being assessed.

4.1.1.2 Primary Consumers

Primary consumers are organisms that occupy the second level in the aquatic food chain. These organisms generally eat plant material such as phytoplankton.

Zooplanktons

Zooplanktons such as *Cladocerans* (a member of the zooplankton group) are found in most aquatic ecosystems in the north. Although *Cladocerans* may be seasonally abundant, their overall contribution to aquatic ecosystems in the north is relatively small. Overall, rotifers were the most

abundant taxonomic group in all of the lakes sampled within the Project area in both the spring and fall.

Benthic Invertebrates

Due to the association of benthic invertebrates with sediments in aquatic ecosystems, they are useful to represent the potential risk to ecosystems from changes in sediment quality. Benthic invertebrates both live and feed within sediments and therefore may be exposed to constituents through ingestion of sediment bound constituents and also through exposure to interstitial waters within the sediment.

Benthic invertebrates provide a link between aquatic and terrestrial ecosystems. *Chironomidae* (midge) larvae are usually the most abundant benthic invertebrate taxa present in aquatic ecosystems in the northern climate. Many species feed on decaying organic matter and thereby form an important link between the decomposer level and primary consumers. Furthermore, midge larvae are a main food source for small/juvenile fish and larger omnivorous fish. The midge adults are capable of flight and are frequently consumed by birds and bats. This life stage provides an important link between aquatic and terrestrial ecosystems in the region.

4.1.1.3 Secondary and Tertiary Consumers

Arctic grayling and lake trout are the most widely dispersed species in the LAA and the species found in the highest abundance (Appendix 5C). Other species (e.g., ninespine stickleback, slimy sculpin and burbot) occur in lesser numbers and their distribution is much more restricted.

Ecological receptors at the secondary consumer level include forage fish, which feed primarily on benthic invertebrates and are an important food source of larger predatory fishes. Examples of fish from the forage category are burbot and white round fish.

Tertiary consumers are found at the top end of the aquatic food chain and consist of larger piscivorous (predatory) fish species such as Arctic grayling, cisco, and lake trout. These predatory fish consume other fish species. Predatory fish are also an important component of the human food chain. Both forage and predatory fish are an important component of the diet of omnivores (e.g., bear).

For the purposes of the risk assessment, specific fish species are not selected for the assessment; rather, general consumer levels are considered. The information from site surveys demonstrates the presence of both forage fish (i.e., burbot) and predatory fish (i.e., lake trout and arctic grayling) in Judge Sissons Lake (Appendix 5C).

4.1.1.4 Summary of Aquatic Receptors

The risks were considered for each of receptors in each of the segments Judge Sissons Lake. Judge Sissons Lake is the largest lake in the area, and forage and predatory fish can potentially exist year-round.

4.1.2 Terrestrial Receptors

Terrestrial receptors selected for this assessment cover all food chain levels in aquatic and terrestrial systems and are listed in Table 4.1-1. Terrestrial receptors were evaluated in various areas of the project area (Figure 3.1-1) and the following sections provide the rationale for the selection.

4.1.2.1 Background

Caribou are perhaps the most significant terrestrial species in the area. Field studies and collaring data have concluded that caribou from several identified herds occur within the Kiggavik RAA in different seasons (Appendix 6C). The five caribou herds identified in Nunavut's mainland region are the Beverly, Ahiak, Wager Bay, Lorillard, and Qamanirjuaq herds. All of these herds are known to occur within the Kiggavik RAA, although at different frequencies and seasons depending on their herd ranges. Although historically, the Kiggavik RAA was identified as a post-calving area for Beverly caribou, current use by this herd during the post-calving season appears to be limited. Some members of the Beverly/Ahiak herd may still be using the Kiggavik RAA during the post-calving period. The Qamanirjuaq herd has not historically been identified as using the Kiggavik RAA during any season; however, in 2009 and 2010, ground observations and collaring data found that most animals occurring within the Kiggavik RAA during late July and early August were from the Qamanirjuaq herd. Animals do not remain in the area long before they move rapidly to the south to wintering areas. Collaring data on the northeast mainland herds (Wager Bay, Lorillard and 'resident' herds) has supported the view that most caribou occurring in the Kiggavik RAA in the winter are individuals from these herds.

Another predominant mammal species in the Kivalliq region is the muskox, which was seen regularly and in herds up to 80 individuals during baseline studies (Appendix 6C). The majority of muskoxen groups were in heath tundra and tussock hummock habitats. Large predatory species, including grizzly bear, wolverine and wolf, are present throughout the Kiggavik RAA but are observed infrequently. Smaller predatory species, such as Arctic fox and ermine, are also present, along with their primary prey such as Arctic hare, sik sik or Arctic ground squirrel, voles and lemmings. (Appendix 6C)

Of terrestrial vertebrates, birds are represented by the greatest number of species. Several raptor species occur, including the peregrine falcon, which nests within the Kiggavik RAA. Other raptor species, including the federally-listed short-eared owl, and rough-legged hawk are known to nest but

at low densities. Raptor species feed primarily on small mammals and upland birds, of which there are several common species in winter (i.e., ptarmigan) and summer (e.g., Lapland longspur, horned lark and savannah sparrow). Waterbirds, including ducks, geese, swans, gulls, terns and shorebirds are also well represented in the Kiggavik RAA. The most common waterbird species are Canada goose, semipalmated sandpiper and long-tailed duck. (Appendix 6C)

The Kiggavik Project area is located in a low arctic region characterized by minimal topography, predominantly shrub tundra habitat, an extreme northern climate, and relatively few terrestrial vertebrates (Appendix 6B). Plant species that are primary forage for herbivorous mammals and birds included fruit and vegetation from blueberry plants (*Vaccinium myrtilloides*), willow (*Salix* spp.), dwarf birch (*Betula nana*), sedge (*Carex* spp.) and lichen.

4.1.2.2 Selection of Terrestrial Receptors and Pathways

Based on the relevant VECs for the project and information on the presence and signifigance of various ecological species in the LAA and RAA, the following terrestrial receptors were selected for the ERA.

Dermal exposure is generally not a significant pathway of exposure for wildlife as fur and feathers are effective at blocking direct contact with skin. A quantitative assessment was not carried out for inhalation of dust by wildife. The exposure from the inhalation route is expected to be much less than from the ingestion pathway; therefore, in general the amount of exposure is considered to be very small and would not in essence significantly affect the predicted exposure. In addition, there is very limited toxicity data with which to assess the inhalation pathway. This scenario does not meet the conditions where inhalation or dermal exposure should be considered based on the British Columbia protocol (SAB CS 2008). Exposure through the dermal and inhalation pathways was thus not assessed quantitatively; this is consistent with the procedure followed by the U.S. EPA (U.S. EPA, 2003, 1999b).

Herbivores

Barren-ground caribou, muskoxen, ptarmigan, Arctic ground squirrel, brown lemming, Lapland longspur, and masked shrew are herbivores and thus are mainly exposed to contaminants via atmospheric deposition on terrestrial vegetation. Herbivores convert vegetable matter to animal protein, and in turn are consumed by omnivores and carnivores. They are also trapped or hunted for fur and food. Herbivores span a wide range of species, including microorganisms, invertebrates and vertebrates.

Figure 4.1-1 provides a schematic diagram of the exposure pathways considered for the herbivores considered as terrestrial ecological receptors for the assessment. The diagram shows an overview of the diet components considered in the assessment for these receptors.

Barren-ground caribou (*Rangifer tarandus groenlandicus*) were selected because they are an important source of food for people in the area (Volume 3). The atmospheric pathway is an important pathway of exposure for caribou since their diet is composed mainly of lichen. Constituent concentrations in lichen are linked directly to constituents in air through direct deposition and nutrient uptake from air. Therefore, the inclusion of caribou is appropriate for assessment of the potential effects on this important regional species from the transfer of constituents through the air pathway. Caribou are also part of the grizzly bear, gray wolf, and wolverine food chain.

Muskoxen (*Ovibos moschatus*) were selected because they have been an integral part of the Inuit lifestyle for centuries as one animal can provide a great amount of meat, a warm versatile hide, and soft insulating fur.

Rock ptarmigans (*Lagopus muta*) were selected because they are occasionally consumed by humans. Ptarmigan are terrestrial birds that are common in northern areas. The ptarmigan will be a surrogate in the assessment for other terrestrial birds with similar diets, such as Canadian goose and grouse.

Arctic ground squirrels (*Sper mophilus parryii*) were selected since they are part of the food chain. They consume browse and herbaceous vegetation. They have a significant role in the food chain, since they are part of the diet of larger predatory species. The Arctic hare is also a common species for the area and was identified as a possible human food source; however, the hare has essentially the same diet as the Arctic ground squirrel. Therefore, exposures to the Arctic hare are captured with the inclusion of the Arctic ground squirrel in the assessment. The Arctic ground squirrel will also serve as a surrogate for Arctic hare in the human diet.

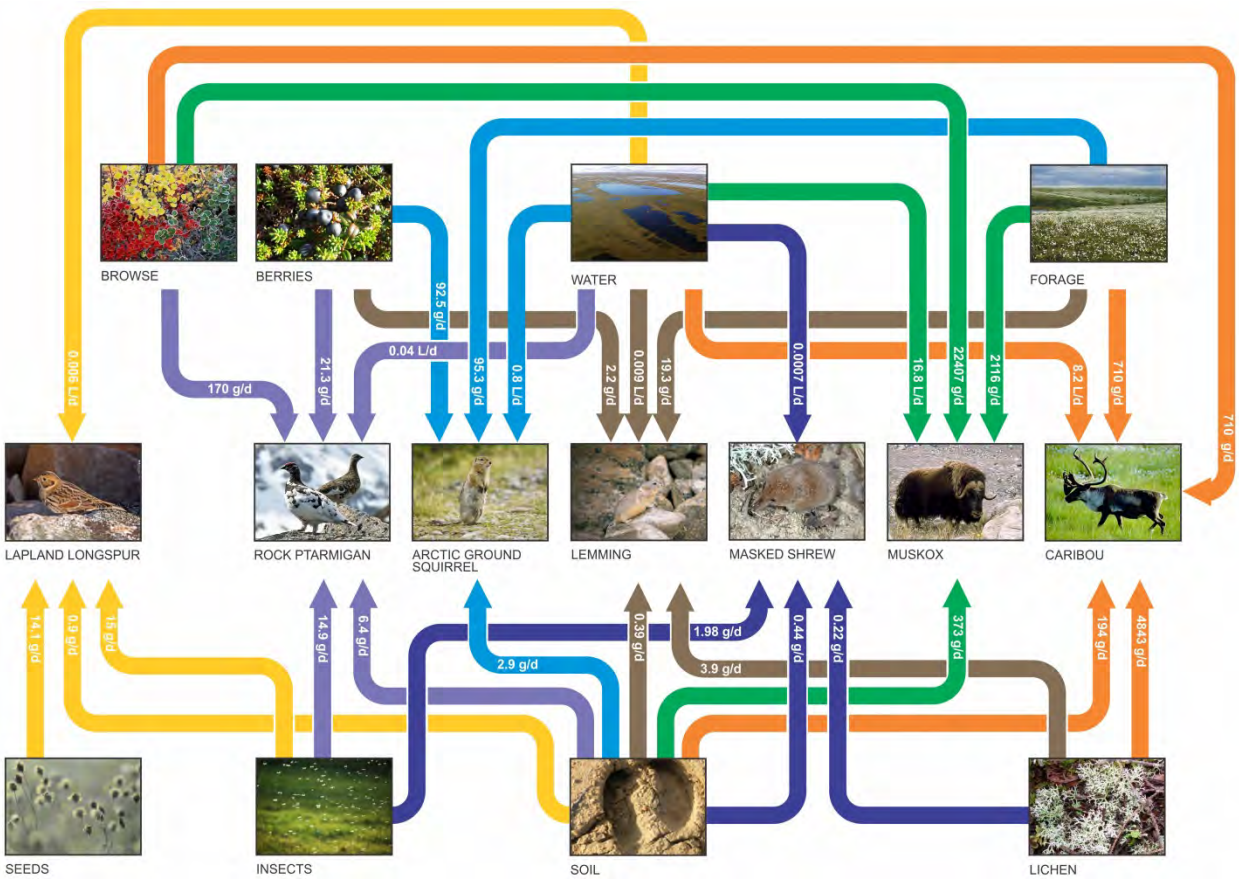


Figure 4.1-1 Schematic of Exposure Pathways for Herbivores

Brown lemmings (*Lemmus sibiricus*) are an important food source for a number of other animals. They consume herbaceous vegetation, berries, and lichen. The lemming acts as a surrogate species for other small herbivore mammals, such as the vole.

Lapland longspurs (*Calcarius lapponicus*) are a common songbird on the Arctic tundra. The longspur was selected as an ecological receptor because its diet consists of seeds and insects. It will represent other small songbirds that might be present in the area.

Masked shrew (*Sorex cinereus*) is a common small mammal that likes riparian and wetland environments. It relies heavily on insects for food, with a small portion of the diet from lichen.

Omnivores

Omnivores consume both plant and animal matter. Vertebrate omnivores included in the assessment include waterfowl and bear. Figure 4.1-2 provides a schematic diagram of the exposure pathways considered for the waterfowl included in the assessment. Figure 4.1-3 provides similar information for

the bear. These diagrams show an overview of the diet components considered in the assessment for these receptors.

Waterfowl selected for the assessment include northern pintail (*Anus acuta*), red-breasted merganser (*Mergus serrator*), semipalmated sandpiper (*Calidris pusilla*), and long-tailed duck (*Clangula hyemalis*). These species, which feed on fish, insects, aquatic vegetation, and aquatic (benthic) invertebrates, cover different diet compositions and food chain effects.

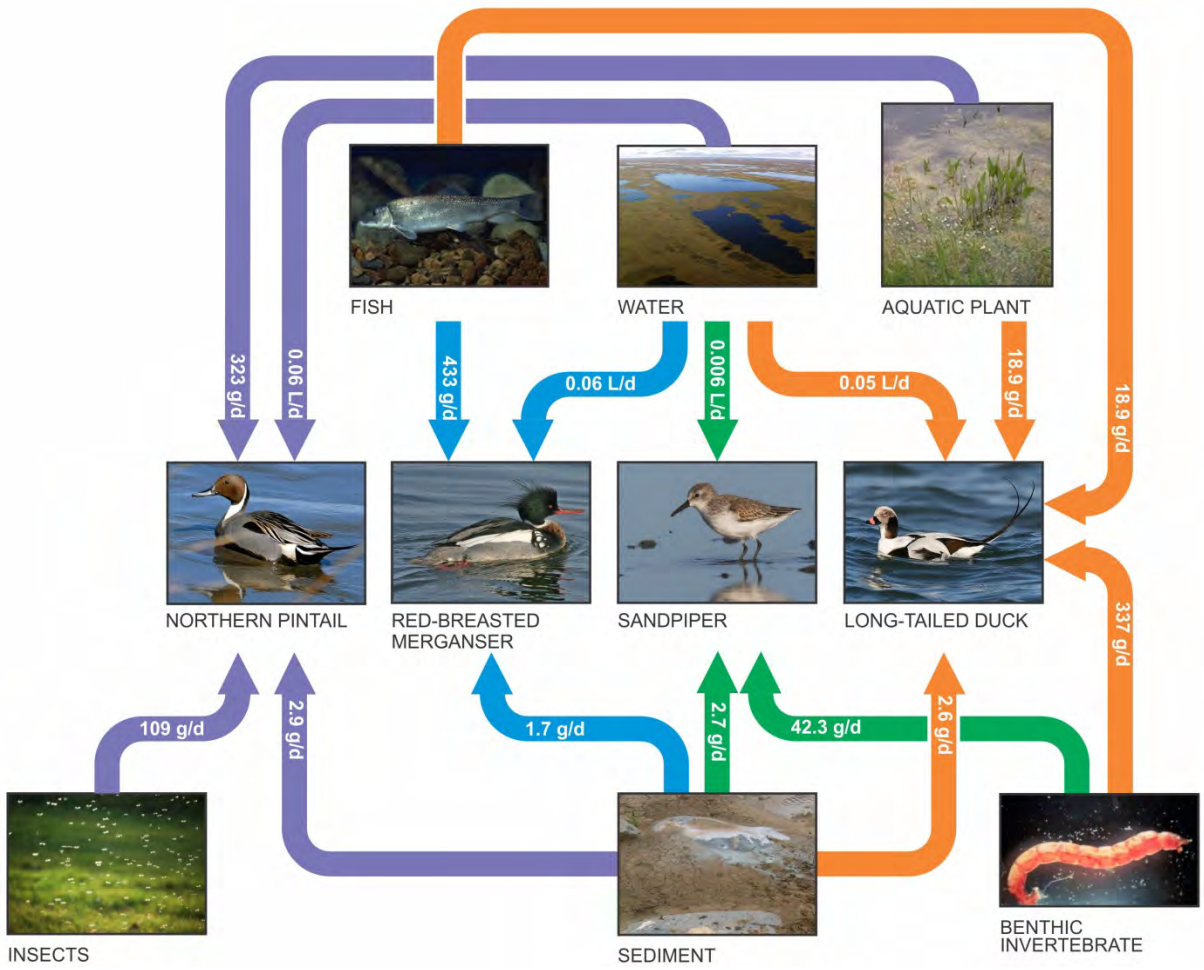


Figure 4.1-2 Schematic of Exposure Pathways for Waterfowl

The diet of the grizzly bear (*Ursus arctos*) is composed of terrestrial vegetation, berries, fish, and caribou, and therefore, the bear is an important indicator of potential effects of atmospheric deposition on the terrestrial environment (and transfer to terrestrial vegetation, berries and soil) as well as emissions to the aquatic environment (and uptake by fish).

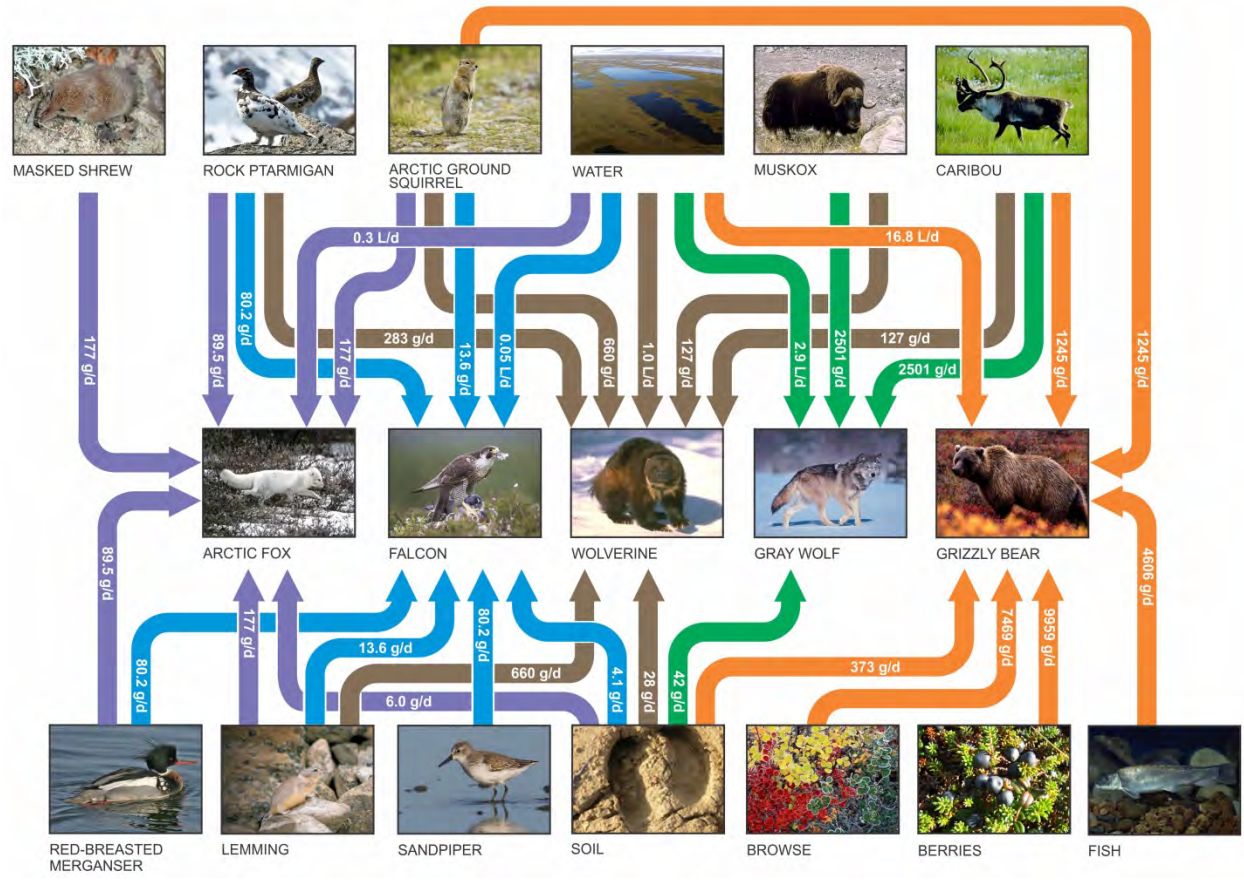


Figure 4.1-3 Schematic of Exposure Pathways for Omnivores and Carnivores

Carnivores

Falcon, fox, wolf and wolverine are predatory species at the top end of the food chain. Predators interact with prey species (usually herbivores) and may influence population levels and distribution of prey. Figure 4.1-3 provides a schematic diagram of the exposure pathways considered for the carnivores included as terrestrial ecological receptors for the assessment. The diagram shows an overview of the diet components considered in the assessment for these receptors.

Peregrine falcons (*Falco peregrinus*) were selected because they are exposed to smaller birds and mammals and are known to nest in the area (Appendix 6C). Exposures to the falcon represent potential exposures to other similar species, such as the short-eared owl. The short-eared owl is also a species of special concern identified within the Sissons Lease area, but it is not specifically considered in the assessment.

Arctic fox (*Alopex lagopus*) were selected since they consume small terrestrial animals and birds and are an important species for trappers in the area. Predicted exposures to the fox will represent exposure to other scavenger mammals.

Gray wolf (*Canis lupus*) was selected since the diet contains caribou and muskox. Wolf dens have been observed within the project area.

Wolverines (*Gulo gulo*) were identified as an important species for the area and are carnivorous scavengers. The wolverine is a member of the weasel family and has the largest home range of carnivores of its size. The diet of the wolverine consists of small mammals and birds and carrion (caribou and muskoxen).

4.1.2.3 Summary of Terrestrial Receptors

The terrestrial receptors chosen for the current assessment are summarized in Table 4.1-1 with the areas where the species were considered to be present. The receptors were selected to represent a wide range of exposure pathways. The rationale for the selection of different terrestrial species was provided above.Attachment D provides a description of the detailed characteristics for the terrestrial receptors considered in this assessment.

The fraction of time assumed to be spent at each location for the receptors is also provided in Table 4.1-1. These assumptions were based on information related to home range and migration behaviours. A conservative approach was taken in setting the values. For example, for the caribou, estimates of time spent in the LAA and RAA was estimated from available satellite and GPS telemetry data from the governments of Nunavut and Northwest Territories. A number of assumptions were made in the evaluation of the radio-collar data:

- Collared animals were accurately identified as members of a particular herd (i.e., they move and live with what is generally accepted as a particular herd);
- Collared animals accurately represent a cross-section of a herd (i.e., as a subset, they sufficiently reflect the home range or extent of an entire herd); and
- Percentage values represent an estimate of the time any one caribou herd spends in the study areas.

Table 4.1-1 Terrestrial Ecological Receptors and Exposure Areas

Study Area	Receptors (Fraction of Time at Location)	
1. Rock Lake Watershed	Lemming (1.0) Longspur (1.0)[a] Ptarmigan (1.0)	Shrew (1.0) Squirrel (1.0)
2. Judge Sissons Lake – vicinity of Kiggavik discharge (JSL-2)	Duck (1.0)[a] Lemming (1.0) Longspur (1.0)[a] Merganser (1.0)[a] Muskox (0.1)	Pintail (1.0)[a] Ptarmigan (1.0) Sandpiper (1.0)[a] Shrew (1.0) Squirrel (1.0)
3. Kiggavik Camp	Lemming (1.0) Longspur (1.0)[a]	Ptarmigan (1.0) Shrew (1.0) Squirrel (1.0)
4. Judge Sissons Lake – vicinity of Sissons discharge (JSL-8)	Duck (1.0)[a] Lemming (1.0) Longspur (1.0)[a] Merganser (1.0)[a] Muskox (0.1)	Pintail (1.0)[a] Ptarmigan (1.0) Sandpiper (1.0)[a] Shrew (1.0) Squirrel (1.0)
5. Local Assessment Area (LAA)	Bear (0.5) Caribou (0.011)[b] Falcon (0.33)	Fox (1.0) Wolf (0.5) Wolverine (0.5)
6. Regional Assessment Area (RAA)	Caribou (0.047)[c] Falcon (0.33)	Wolf (1.0) Wolverine (1.0)

Note: for a map of areas, please see Figure 3.1-1.

The value in (-) indicates the fraction of the year each receptor is assumed to spend in the study area.

a – migratory birds likely spend only about 4 months of the year in the area of the site; however, fraction of time at each location was assumed to be 1 in order to capture potential effects from 4 months of continuous exposure at the site

b – the fraction of time for the caribou in the LAA is on a triangular distribution T(0.005, 0.011, 0.03).

c – the fraction of time for the caribou in the RAA is on a triangular distribution T(0.003, 0.047, 0.13)

Table 4.1-2 summarizes the results of the analysis of the telemetry data for the caribou herds within the LAA and RAA. The percent occurrences range from 0.0% to 2.9% within the LAA and 0.0% to 13.3% for the RAA. A triangular distribution was used with the minimum of the distribution set to the maximum of the minimum % time in the area result from the telemetry data, the mode of the distribution was set to the maximum average % time in the area, and the maximum was set to the maximum maximum % time in the area.

Table 4.1-2 Caribou Residency Time Based on Telemetry Data

Statistic	Percent Occurrence	
	LAA	RAA
Maximum Minimum	0.5%	0.3%
Maximum Average	1.1%	4.7%
Maximum Maximum	2.9%	13.3%
Note: Additional information on how these values were derived from the collar data is included in Volume 6		

Finally, the ingestion of tailings by caribou was not considered in the quantitative assessment for exposure of caribou from the site. A quantitative assessment is not necessary since the comprehensive in-pit tailings management strategy will minimize this risk. Even in a delayed post-closure scenario, site presence would be maintained and appropriate mitigation measures would be implemented.

4.2 Human Receptors

To cover a variety of potential exposure situations from the site, both members of the public and workers were included in this assessment. These include permanent residents of the Baker Lake community, hunters at Judge Sissons Lake and non-nuclear workers at the project site.

4.2.1.1 Selection of Human Receptors

Members of the public that were identified for inclusion in the risk assessment were focused on adult, child and toddler family members in Baker Lake, the closest community to the Kiggavik Project. As Baker Lake is located 75 km away from the proposed mine site and other communities are situated at even greater distances, it was decided to limit the assessment to Baker Lake. Potential effects in other communities would be expected to be less than assessed for Baker Lake residents. Chesterfield Inlet and Rankin Inlet, the next closest communities, are both located about 350 km from the Kiggavik site.

Baker Lake is located at almost the geographic centre of Canada; 940 km east of Yellowknife and 256 km northwest of Rankin Inlet. Most of the residents are descendants of the Thule Inuit who migrated to Keewatin about 1,000 years ago. It is the only inland community in Nunavut. Caribou and inland fish are important components of the local diet. Traditional activity remains highly valued and the number of hunters appears stable (Volume 9).

The hunter at Judge Sissons Lake was selected based on evidence of historical fishing and hunting activities in the area (Volume 3). This receptor is assumed to spend four months at a cabin at Judge Sissons Lake and eight months at the Baker Lake community. Since there is a strong desire to pass

traditional knowledge to the younger generation (Volume 3), a child in the hunter's family was also considered to spend 1 month of the year with the hunter. The remainder of the year, the child is at home in Baker Lake, consuming the food obtained by the hunter at Judge Sissons Lake. A toddler in the family is also considered at hunter's home in Baker Lake.

The worker considered is a non-Nuclear Energy Worker (NEW) who lives at the campsite fifty percent of the time. The worker is assumed to have been hired from the Baker Lake community, where the remaining fifty percent of their time is assumed to be spent. Note, NEW are not considered in the pathways analysis and human health risk assessment as their exposure pathways are quite different from members of the public and therefore human health risks to NEW are assessed separately (see Volume 8).

4.2.1.2 Potential Pathways of Exposure for Human Receptors

Table 4.1-2 summarzies the pathways that were considered for human receptors from the site and the rationale for whether the pathway was included or excluded. The list of pathways was obtained from documents such as the Health Canada Guidance on Human Health Detailed Quantitative Radiological Risk Assessment ($DQRA_{RAD}$), the as well as CSA N288.6 Environmental Risk Assessments at Class I Nuclear Facilities and Uranium Mines and Mills. Several pathways were determined to be insignificant and were excluded from the assessment.

Table 4.2-1 Potential Pathways of Exposure for Human Receptors

Potential Pathway of Exposure	Members of the Public			Comment
	Baker Lake Resident	Hunter at Judge Sissons Lake	Non-NEW*	
Incidental Soil Ingestion	Yes	Yes	Yes	Radionuclides and non-radionuclides can be deposited on soil. Therefore, the incremental exposure from incidental ingestion of soil is a potential pathway of exposure.
Dermal contact with soil	Min	Min	Min	Dermal exposure is expected to be minimal due to the short season where there is the potential for exposed skin.
Inhalation	Yes	Yes	Yes	Inhalation of particulate containing radionuclides and metals as well as inhalation of radon are potential pathways of exposure.
Immersion in air (dermal exposure)	Min	Min	Min	The dermal exposure to air and external dose from radionuclides in air is not expected to be a significant source of exposure.

Table 4.2-1 Potential Pathways of Exposure for Human Receptors

Potential Pathway of Exposure	Members of the Public			Comment
	Baker Lake Resident	Hunter at Judge Sissons Lake	Non-NEW*	
Drinking water	Min	Yes	No	Baker Lake obtains drinking water from Baker Lake which is 75km downstream and the impact of discharge from the site will be insignificant at this location. A hunter at Judge Sissons Lake is assumed to consume water from this lake while in the area. Potable water for the site will be taken from Mushroom and Siamese lakes. The water quality in these lakes will not be impacted by the Project.
Other uses of potable water (e.g., bathing)	Min	Min	No	There will be no significant change in water quality in Baker Lake and water for the camp will be sourced from a un-impacted water body. There is expected to be minimal exposure through secondary exposure pathways of water, such as bathing, for the hunter at Judge Sissons Lake.
Harvest local foods (e.g., berries)	Yes	Yes	Yes	IQ supports that berries are important and it was assumed all receptors would gather berries.
Hunting / Trapping	Yes	Yes	No	Caribou is an important food source. While on site the non-NEW will not consume caribou from local sources.
Fishing	Min	Yes	No	Fish is an important food source. There will be no significant change to water quality in Baker Lake; therefore, fish obtained from this source will not result in any significant exposure. The hunter at Judge Sissons Lake will obtain fish from this lake and take it back for consumption throughout the year by all members of the family. While on site the non-NEW will not consume fish from local sources.
Garden produce ingestion	Min	Min	No	Due to the short growing season it was assumed that there would not be any significant exposure to produce grown in backyard gardens.
Irrigation of vegetation (potable / groundwater / surface water)	No	No	No	Any irrigation water would be sourced from Baker Lake, which is assumed to not have any significance impact. Combined with the low preponderance of gardens this route of exposure is insignificant.
Livestock	No	No	No	No domestic livestock in the area.
External dose from soil	Yes	Yes	Yes	The incremental external dose from radionuclides deposited on soil is a potential pathway of exposure.
Recreational use of surface water (e.g., swimming[(a)])	Min	Min	No	Due to the climate, it is not expected that there would be any significant use of Judge Sissons Lake for recreational purposes such as swimming or beach.

Table 4.2-1 Potential Pathways of Exposure for Human Receptors

Potential Pathway of Exposure	Members of the Public			Comment
	Baker Lake Resident	Hunter at Judge Sissons Lake	Non-NEW*	
Note: Yes Potential Pathway No Not potential pathway of exposure Min Minimal exposure, not significant and will not be quantified. Professional judgment utilized in decision making process. * The non-NEW is assumed to reside in Baker Lake and work for half the time at the site. The pathways identified in this table are for the time spent at the site. a Includes the pathways of incidental surface water ingestion, immersion in water (dermal exposure and external dose), incidental sediment ingestion, external dose from sediment, dermal contact with sediment				

4.2.1.3 Dietary Characteristics of Human Receptors

The human diet derived for the assessment was based on a survey of the present day diet of the Kivalliq community (Kuhnlein et al. 2000) as well as the Inuit Qaujimajatuqangit (IQ) for Baker Lake (Appendix 3B). Kivalliq, located on the western shores of Hudson's Bay, is home to seven communities including the Baker Lake and Chesterfield communities.

Based on the IQ, caribou and fish were consistently named the primary food sources. Trout and whitefish were cited as common fish. Squirrel was identified by one interviewee as a food source, but only in the case of the lack of other foods.

Kuhnlein et al. (2000) identified the most commonly reported foods: caribou, fish (arctic char, salmon, cod, trout, and whitefish), muskox, grouse/ptarmigan, hare, berries (cranberry and blackberry), and duck. Some marine species, such as seal, whale, polar bear, and walrus, were identified in Kuhnlein et al. (2000), but these species were not considered for the assessment as only 2% of Baker Lake households reported harvesting sea mammals (Appendix 9A). It is noted that almost 40% of people reported that they ate at least some sea mammals in 2010 but is this likely due to intra community sharing of harvests, as well as some limited commercialization. Table 4.2-2 summarizes the winter season consumption of traditional food in Baker Lake. From Table 4.2-2, it is indicated that caribou and fish are the primary traditional foods consumed during the winter season.

Table 4.2-2 Winter Season Consumption of Traditional Food, in Baker Lake

	% of People Eating 2010	Frequency (rating*)	
		2010	1998/99
Sea Mammals	39		
Beluga	34	VL	VL
Narwhal	13	VL	VL
Walrus	5	VL	VL
Bowhead	3	VL	0
Ringed Seal	7	VL	VL
Bearded Seal	2	VL	0
Harp Seal	1	VL	VL
Harbour Seal	3	VL	0
Mammals	99		
Polar Bear	1	VL	0
Caribou	99	H	H
Musk Ox	10	VL	VL
Birds	14		
Ducks	3	VL	0
Geese	7	VL	VL
Ptarmigans	8	VL	VL
Gulls and Terns	1	VL	0
Fish	78		
Trout	63	L	VL
Whitefish	21	VL	VL
Burbot	2	VL	0
Greyling	5	VL	VL
Char	60	M	L
Shellfish	4	VL	0

Note: * VL = consumed on < 3 days over the three month winter season December to February, L = 3 to 6 days/season, M = 6 to 12 days/season and H = 12 to 24 days/season.
Sources: AREVA survey for 2010 data; Kuhnlein et al, 2000 for 1998/99 data.

Muskox and duck ingestion were not included in the assessement for human receptors. These foods were identified in Kuhnlein et al. (2000), but not in the IQ for Baker Lake. Kuhnlein et al. (2000) considered a number of communities, including Baker Lake, but not all foods identified would necessarily apply to all communities. One IQ participant referred to the protection of muskoxen and an established sanctuary around the Thelon area as prohibiting the use of muskox for food (IQ BL02 2008). Consideration of caribou and other herbivore mammals (such as squirrel) in the human diet for the assessment would include any potential exposures from ingestion of muskox. Similarly, the ingestion of waterfowl was not indicated in the IQ for Baker Lake, except as mentioned by one or two individuals as the rare consumption of geese (geese generally consume terrestrial vegetation).

The ingestion of squirrel was considered in the assessment because squirrels were mentioned as a potential food source in the IQ for Baker Lake (Appendix 3B) and hare (similar dietary characteristics to squirrel) was indicated as a potential food source in Kuhnlein et al. (2000). Therefore, a small part of the human diet was attributed to the ingestion of squirrel (which also serves as a surrogate for other small mammals). A small portion of the diet was also assigned to ptarmigan. This source was identified by Kuhnlein et al. (2000), although expected to comprise a small portion of the diet. The recent survey of Baker Lake was consistent with this information (Appendix 9A).

The IQ studies completed as part of the baseline program discussed the use of certain plant species as food, medicine, shelter, and other human uses (see Appendix 3B). Cowberries, blueberries, cloudberries, ‘black’ berries and ‘red’ berries were gathered in the past and are still used today. Some plants (e.g., cloudberry leaves, tundra moss) are used to make tea. Elders commented during focus groups that traditional cures were no longer used. IQ interviews did not identify particular places for collecting plants; rather it was noted that plants were everywhere. The area around Sissons Lake was noted to be particularly good for red berries. Plants are typically gathered by Elders from August to September. Based on survey results from Baker Lake (Appendix 9A), 40% of households are engaged in plant collecting, although most households (87%) are likely not engaged regularly (one a day or two every now and then, likely during day trips and weekends). The study noted that this type of occasional involvement did not reflect a low level of commitment or harvesting. Almost seven percent of households indicated they had purchased wild berries over the course of the year.

Figure 4.2-1 illustrates the exposure pathways for human receptors considered in the assessment. Table 4.2-3 summarizes the dietary characteristics assumed for the human receptors. The assessment has been completed in a probabilistic framework and the distributions used to represent the uncertainty in the assumptions are provided in the table.

Table 4.2-3 Human Receptor Dietary Characteristics for the Kiggavik Project

	Units	Receptor		
		Adult	Child	Toddler
Water Intake	m^3/yr	LN (0.6, 1.65, 0.22, 1.6)	LN (0.3, 1.49, 0.14, 0.67)	LN (0.22, 1.49, 0.10, 0.50)
Soil Intake	mg/d	LN (1.7, 2.9, 0, 68)	LN (12, 27, 0, 783)	LN (13, 18, 0, 283)
Total Meat Intake	g (ww)/d	LN (350, 2.5, 56, 2190)	LN (150, 2.5, 24, 938)	LN (100, 2.5, 16, 625)
Fraction Fish Intake	(-)	U (0.23, 0.36)*	U (0.23, 0.36)*	U (0.23, 0.36)*
Fraction Ptarmigan Intake	(-)	U (0.02, 0.06)	U (0.02, 0.06)	U (0.02, 0.06)
Fraction Squirrel Intake	(-)	U (0.02, 0.06)	U (0.02, 0.06)	U (0.02, 0.06)
Fraction Caribou Intake	(-)	U (0.60, 0.65)	U (0.60, 0.65)	U (0.60, 0.65)
Berry Intake	g (ww)/d	LN (1.8, 3, 0.2, 16.2)	LN (8.5, 3, 0.9, 77)	LN (6.4, 3, 0.7, 58)
Medicinal Tea Intake	L/d	T (0.023, 0.043, 0.085)	C (0)	C (0)
Inhalation Rate	m^3/d	LN (16.6, 4.1, 9.5, 33)	LN (14.5, 3.4, 8.3, 25)	LN (8.3, 2.2, 4.6, 15.6)
Body Weight	kg	C (70.7)	C (32.9)	C (16.5)

Note: These intake amounts only consider the portion of the diet from local food sources.

C = Constant distribution (value) T = Triangular distribution (Min, Mode, Max) LN = Lognormal distribution (Geomean, Geo Std Dev, Min, Max) U = Uniform distribution (Min, Max)

* calculated as remainder of ptarmigan, squirrel, and caribou fractions in the LAKEVIEW/INTAKE code.

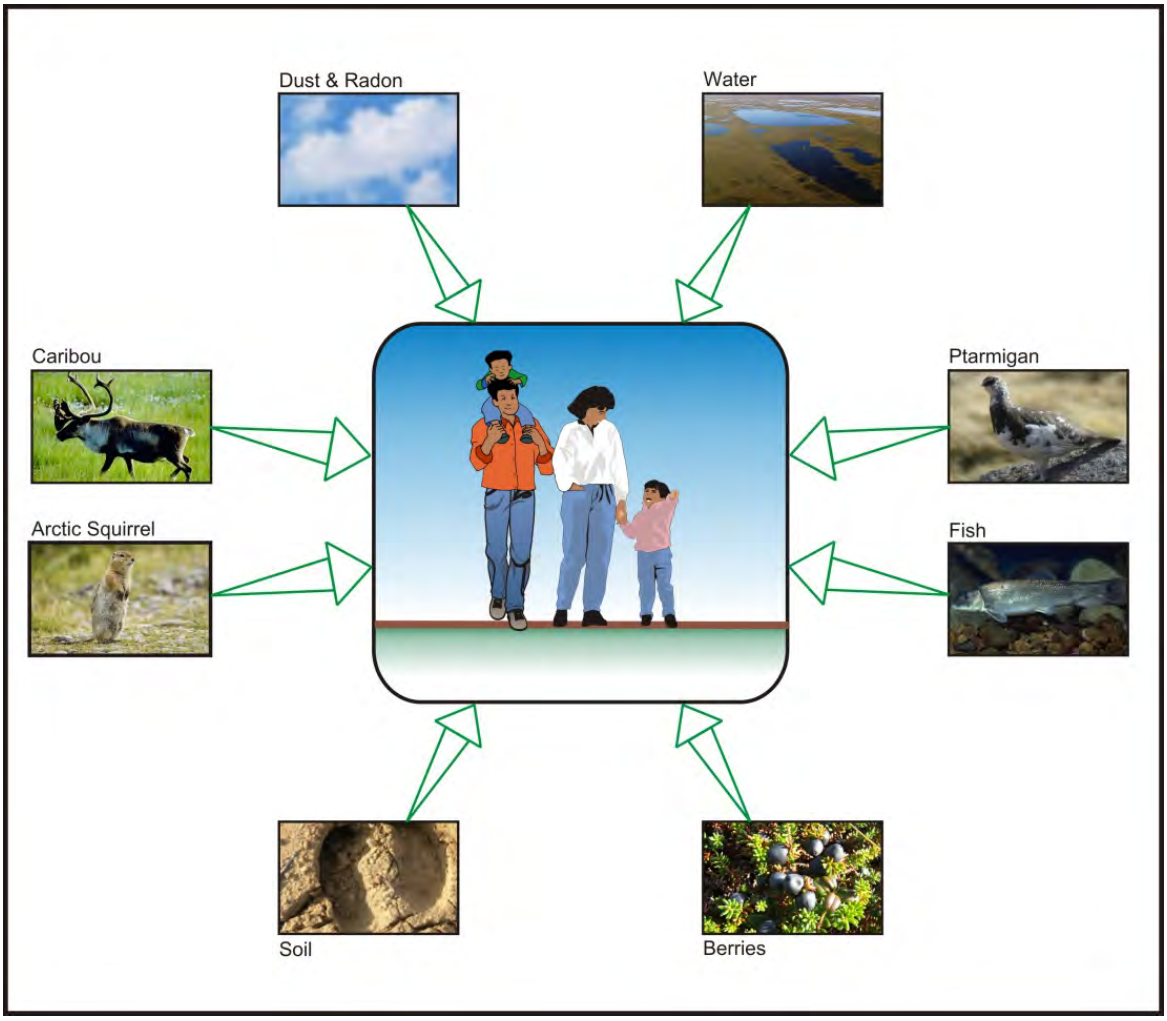


Figure 4.2-1 Schematic Representation of Exposure Pathways for Human Receptors

Water Intake

A lognormal distribution was assumed for drinking water intake rate, using mean and standard deviation values reported in Richardson (1997) *"Compendium of Canadian Human Exposure Factors for Risk Assessment"*. Maximum values were based on the 90th percentile from recent unpublished data analyzed by Health Canada (Richardson 2009a) and minimum values were assumed.

Soil Intake

The Contaminated Sites Division (CSD) of Health Canada has undertaken to re-evaluate soil ingestion rates and to define these as lognormal probability distribution functions (PDFs). Numerous studies have been conducted to quantify soil ingestion rate (reviewed by Doyle *et al*. 2010). Doyle *et al*. (2010) has indicated that the quantities of soil ingested, as deduced from those studies, are as much an artifact of limited study statistical strength and minimum analytical detection limits, than related to true quantification of actual soil ingestion.

It is broadly acknowledged among professional risk assessors that the assumed rates of soil ingestion currently used for risk assessment significantly over-estimate actual exposure from this environmental medium. Also, the assumptions concerning soil ingestion currently employed and recommended for risk assessment by Health Canada (2004a) are not associated in any way with the amount of time spent in the outdoor environment. As a result, the same assumed rate of soil ingestion is applied whether a hypothetical receptor is assumed to be outdoors for 10 minutes or 10 hours. However, with the exception of children exhibiting pica behaviour, it is logical to expect that the amount of soil ingested, and certainly the likelihood that soil ingestion would actually occur, would be greater if that receptor were outdoors 10 hours versus 10 minutes.

Based on methods originally developed for estimation of indoor settled dust ingestion rates (WTCWG 2003), and accounting for various factors such as time spent outdoors, soil adherence to hands, rates of hand-to-mouth activity, surface area of hands, etc., the CSD (unpublished) has developed the lognormal PDFs for soil ingestion rate presented in Table 4.2-2.

Air Intake

A lognormal distribution was assumed for inhalation rate. Mean and standard deviation were based on Allan *et al*. (2008). The 5th and 95th percentiles were used as minimum and maximum. These values were not reported in Allan et al. (2008) but were provided separately by Richardson (2009b).

Body Weight

The body weight (bw) of a toddler, child and adult are also necessary in order to calculate a daily intake (mg/(kg (bw) d)). In this assessment, the body weights used for the toddler, child and adult receptors were 16.5 kg, 32.9 kg and 70.7 kg, respectively (Richardson 1997). Body weights were held constant in the assessment.

Medicinal Tea Intake

Kuhnlein et al. (2000) did not include medicinal tea in the regional frequency questionnaire. The IQ for Baker Lake (Appendix 3B) mentions of the ingestion of tea, generally, throughout the interviews.

In the absence of data on consumption amount, the Yukon dietary survey (Receveur et al. 1998) indicated that in the winter 12% of the population consumes Labrador Tea 1.2 days/week and in the summer 23% of the population consumes Labrador tea 1.4 days/week. This equates to an average Labrador tea consumption of 1.3 days per week. The survey does not provide the amount of Labrador tea or other medicinal teas that are consumed.

To estimate the amount of medicinal tea consumed by members of the community, a web search was undertaken. The web search indicated that "*Labrador Tea contains small amounts of the toxin andromedotoxin which can cause headaches, cramps, paralysis and intestinal problems if too much is consumed. As a general rule, this tea should be consumed in moderation. One cup is often considered the safe amount.*" - *http://www.laurentiancenter.com/plantkey/plants/labradortea.html*

Therefore, it was assumed in this assessment that an adult (70 kg body weight) would consume 250 mL (1 cup) of medicinal teas for 1.3 days a week. This equates to a consumption rate of 0.17 cups/day. There was no information available on the amount of tea used in the brewing process. In this assessment, it was assumed that approximately 3 g were used. This is equivalent to a bag of regular tea. A triangular distribution was established around this value with an assumed range of +/- a factor of 2 to capture a range of possible consumptions.

Food Intake

The food ingestion rates for local food developed from Kuhnlein et al. (2000) in were used for the current assessment. Lognormal distributions were assigned with a geometric standard deviation of 2.5 around a geometric mean value to obtain an arithmetic mean value consistent with the nominal value for the total meat ingestion rate (including caribou, ptarmigan, squirrel, and fish). Similar distributions were developed for ingestion rates of berries, with nominal values based on the Hatchet Lake Band survey (CanNorth 2000) and a geometric standard deviation of 3.0. The LAKEVIEW/INTAKE code considers the fraction of meat ingestion for each component (i.e., caribou)

and therefore, the information is presented this way in Table 4.2-3. Uniform distributions were derived with some accommodation for the squirrel portion of the diet coming from the allotment for caribou. The fraction of the diet that is fish is calculated as the remainder of the diet in the LAKEVIEW/INTAKE code; however, the values presented in Table 4.2-3 represent the likely distribution for the fish portion of the diet based on the other distributions. The distributions are intended to account for the variation in diet among individuals but is not meant to represent any one specific person.

A study for the Beverly and Qamanirjuaq Caribou Management Board (BQCMB) (InterGroup 2008) was also considered. This report indicates that 25 caribou are harvested per domestic hunter and that each caribou has an edible weight of 45 kg. This would correspond to 1125 kg of caribou meat in a year for a hunter. Based on the intake rates provided in Table 4.2-1, one hunter can support at least 10 individuals.

4.2.1.4 Summary of Human Receptors

Table 4.2-4 summarizes the assumptions for the exposure pathways and locations for the human receptors considered in the assessment. The table shows that a variety of potential exposures are accounted for in the assessment. The hunter is assumed to be a resident of Baker Lake and spent 4 months at Judge Sissons Lake. During this time caribou, fish, small game and berries are harvested and consumed while back at the primary residence and shared with other members of the family. Although it is not expected that children would accompany the adult hunter every trip, the transfer of traditional ways is important and thus children were assumed to accompany the hunter one month each year.

Table 4.2-4 Exposure Pathways and Locations Considered for Human Receptors

Pathway	Non-NEW at Campsite	Hunter at Judge Sissons			Baker Lake Residents
		Adult	Child	Toddler	
Inhalation	0.5 Campsite (3) 0.5 Baker Lake (7)	0.33 JSL (2) 0.67 Baker Lake (7)	0.08 JSL (2) 0.92 Baker Lake (7)	1.0 Baker Lake (7)	1.0 Baker Lake (7)
Groundshine	0.5 Campsite (3) 0.5 Baker Lake (7)	0.33 JSL (2) 0.67 Baker Lake (7)	0.08 JSL (2) 0.92 Baker Lake (7)	1.0 Baker Lake (7)	1.0 Baker Lake (7)
Drinking Water	1.0 Background	0.33 Average of JSL-1 and JSL-2 0.67 Background	0.08 Average of JSL-1 and JSL-2 0.92 Background	1.0 Background	1.0 Background
Small Game	0.0	1.0 JSL (2)	1.0 JSL (2)	1.0 JSL (2)	1.0 Baker Lake (7)
Large Game (Caribou)	0.0	1.0 Local Assessment Area (5)	1.0 Local Assessment Area (5)	1.0 Local Assessment Area (5)	1.0 Regional Assessment Area (6)
Fish	0.0	1.0 Average of JSL-1 and JSL-2	1.0 Average of JSL-1 and JSL-2	1.0 Average of JSL-1 and JSL-2	1.0 Background

Table 4.2-4 Exposure Pathways and Locations Considered for Human Receptors

Pathway	Non-NEW at Campsite	Hunter at Judge Sissons			Baker Lake Residents
		Adult	Child	Toddler	
Berries	0.5 Campsite (3) 0.5 Baker Lake (7)	1.0 JSL (2)	1.0 JSL (2)	1.0 JSL (2)	1.0 Baker Lake (7)
Soil	0.5 Campsite (3) 0.5 Baker Lake (7)	0.33 JSL (2) 0.67 Baker Lake (7)	0.08 JSL (2) 0.92 Baker Lake (7)	1.0 Baker Lake (7)	1.0 Baker Lake (7)

Note: 0.5 Campsite (3) represents fraction of time (0.5) at location Campsite (area 3). Refer to Figure 3.1-1 for locations.

The non-NEW worker at the Project Site will not be allowed to hunt or fish while working; therefore, these ingestion pathways are not considered for the worker.

The adult hunter is assumed to spend 4 months of the year hunting in the vicinity of Judge Sissons Lake. A child is assumed to join the hunter for 1 month of the year, while a toddler is assumed to spend all of the year at home in Baker Lake. The adult, child, and toddler all eat the food obtained by the hunter from Judge Sissons Lake area.

4.3 Overview of Receptors and VECs/VSECs

The entire list of receptors considered in this assessment is provided in Table 4.3-1, along with the VEC/VSEC that they represent.

Table 4.3-1 Summary of Receptors and VEC / VSEC

VEC / VSEC	Receptor
Terrestrial environment, including soils	• Soil quality
Surface water quality	• Surface water quality
Sediment quality	• Sediment quality
Freshwater aquatic environment - aquatic biota (including fish, aquatic macrophytes, benthic invertebrates, and other aquatic organisms)	• Phytoplankton • Zooplankton • Aquatic plants • Benthic invertebrates • Forage fish • Predatory fish
Vegetation	• Lichen • Browse (e.g., willow, birch) • Forage (e.g., sedges) • Berries
Terrestrial wildlife, including representative terrestrial mammals (i.e., caribou, muskoxen, wolverine, grizzly bears, wolves and less conspicuous species that may be maximally exposed to contaminants)	• Barren-ground caribou • Muskox • Grizzly bear • Arctic fox • Gray wolf • Wolverine • Brown lemming

Table 4.3-1 Summary of Receptors and VEC / VSEC

VEC / VSEC	Receptor
	• Masked shrew • Arctic ground squirrel
Birds including raptors, migratory birds and seabirds	• Peregrine falcon • Long-tailed duck • Lapland longspur • Red-breasted merganser • Northern pintail • Rock ptarmigan • Semipalmated sandpiper
Health of public	• Non-NEW at Kiggavik camp • Resident of Baker Lake • Hunter at cabin on Judge Sissons Lake

5 Exposure Assessment

This section discusses the estimated exposures of selected receptors to effects from the Kiggavik Project.

5.1 Transfer Factors

The INTAKE model relies on transfer factors to estimate concentrations in environmental media. Transfer factors are empirical values that provide a measure of the partitioning behaviour of a COPC between two environmental media. Transfer factors can describe partitioning between many different media, including water-to-fish, water-to-benthic invertebrates, food-to-animal flesh and other media.

5.1.1 Aquatic Model Transfer Factors

Section 2.7 provided an overview of the aquatic model. Site-specific Kd_{sed} values were developed from solid phase concentrations, porewater concentrations, and extractable fraction information available from the six-step Tessier extraction test (introduced in Section 3.3). The Kd_{sed} value for each contaminant was evaluated as follows in equation (5.1-1):

$$Kd_{sed} = \frac{f_i \sigma_i}{C_{P,i}} \quad (5.1\text{-}1)$$

Where:

i = specific constituent “i”
Kd_{sed} = sorption (distribution) coefficient (m^3 / kg)
fi = fraction of “exchangeable” constituent “i” in the sediment
σi = total concentration of constituent “i” in the sediment (g/kg)
Cp,i = porewater concentration of dissolved constituent “i” (g/m^3)

The calculated Kd_{sed} values are summarized in Table 5.1-1. It is apparent from Table 5.1-1 that the cobalt concentration in the porewater is solubility rather than sorption controlled. A Kd value above 20 m^3/kg (20,000 ml/g) is indicative of precipitation or co-precipitation rather than sorption equilibrium. The relatively large sorption coefficient for arsenic (Kd = 6.1×10^{-3} m^3/g) is characteristic of ferric hydroxide containing sediments. The affinity of arsenate for iron hydroxide surfaces is well established (Banerjee et al., 2008) and filtration through granular ferric hydroxide columns is often used to remove dissolved arsenic from contaminated groundwater. Nickel sorption is pH dependent and is expected to be greater than 1.0×10^{-3} m^3/g reaching a maximum value at pH= 7.6 (Butow et al., 1990). The relatively high Kd value for nickel in Table 5.1-1, however, is within the reported maximum value of 7.4×10^{-3} m^3/g in sandy aquifers (Christensen et al., 1998).

Table 5.1-1 Summary of Calculated Site-Specific Sorption Coefficients

COPC	Solid-Phase Concentration		Porewater Concentration		Kd_{sed} (m^3/g dw)
Uranium	0.317	µg/g dw	$8.9x10^{-5}$	g/m^3	$1.6x10^{-3}$
Thorium-230	0.043	Bq/g dw	9.22	Bq/m^3	$3.9x10^{-3}$
Lead-210	0.103	Bq/g dw	16.8	Bq/m^3	$3.9x10^{-3}$
Radium-226	0.058	Bq/g dw	5.04	Bq/m^3	$3.9x10^{-3}$
Polonium-210	0.176	Bq/g dw	8.41	Bq/m^3	$1.1x10^{-3}$
Arsenic	8.24	µg/g dw	$2.4x10^{-4}$	g/m^3	$6.1x10^{-3}$
Cadmium	Not calculated				
Cobalt	8.97	µg/g dw	$2.5x10^{-5}$	g/m^3	0.110
Copper	12.9	µg/g dw	$1.3x10^{-3}$	g/m^3	$1.6x10^{-3}$
Lead	15.3	µg/g dw	$9.2x10^{-4}$	g/m^3	$3.9x10^{-3}$
Molybdenum	1.31	µg/g dw	$1.5x10^{-4}$	g/m^3	$1.3x10^{-3}$
Nickel	21.0	µg/g dw	$7.9x10^{-4}$	g/m^3	$4.2x10^{-3}$
Selenium	0.509	µg/g dw	$5.0x10^{-5}$	g/m^3	$2.5x10^{-3}$
Zinc	64.7	µg/g dw	$2.0x10^{-2}$	g/m^3	$1.5x10^{-3}$

The radioisotope lead-210 forms an ideal solid solution with the parent element, lead. Consequently, the Kd values for lead and lead-210 should be identical. Lead sorption in various soils has been extensively studied world-wide. It is reportedly influenced by sediment constituents (clays, oxides, hydroxides, oxyhydroxides, and organic matter) lead concentration and the pH. At a pH range of 6.4 to 8.7 the expected Kd ranges from $1.9x10^{-3}$ to $23.7x10^{-3}$ m^3/g at dissolved lead concentrations of 1 µg/l or less. This compares with an estimated Kd of $3.9x10^{-3}$ m^3/g in Judge Sissons Lake sediments.

Selenium is most likely to exist as selenate (SeO_4^{2-}) or selenite (SeO_3^{2-}) in the top layers of sediments in equilibrium with the water column. Chemically, selenium resembles sulphate and consequently the sorption coefficient is expected to be relatively low, less than $1x10^{-3}$ m^3/g (U.S. EPA, 1999a). By contrast the estimated value reported in Table 5.1-1 is $2.5x10^{-3}$ m^3/g. The reason for the high Kd value for selenium is not clear. Selenium is not expected to be reduced to selenide and precipitate under normal environmental conditions. It may be, that porewater concentrations of selenium were under-predicted during the simulation of porewater to surface water distributions resulting in higher than expected Kd values.

Under oxidizing conditions uranium is expected to exist primarily as the divalent uranyl cation, UO_2^{2+}, in the top sediment layer. Although the uranyl ion is known to form neutral complexes with ligands such as sulphate, carbonate, or chloride, these are not expected to be present in significant concentrations. Depending on the clay and organics content of the sediment, the Kd value for uranium had been reported to vary widely between $0.063x10^{-3}$ to $63x10^{-3}$ m^3/g at pH = 7.0. The upper value most likely corresponds to precipitated U(VI) compounds, since some researchers do

not differentiate between adsorbed and precipitated uranium. The estimated Kd value for uranium in the present study is in line with expectations (Curtis et al., 2004).

It is also noteworthy, that divalent cations such as copper and zinc were estimated to have essentially the same Kd values as uranium. The sorption coefficients for copper and zinc are higher than those normally observed in sandy sediments (Hassan et al., 1996). The higher Kd values are attributed to the elevated iron and aluminum oxide levels in the Judge Sissons Lake sediments. Iron and aluminum oxides and hydroxides are known to provide a good matrix for adsorbing divalent cations (U.S. EPA, 1999a); the affinity of the iron oxide surface for copper and zinc may be two orders of magnitude higher than the common clay mineral surfaces (Takematsu, 1979).

Between pH of 5 to 8, thorium is expected to have a sorption coefficient between $1.7x10^{-3}$ and $170x10^{-3}$ m^3/g (U.S. EPA, 1999a). As in the case of lead and lead-210, thorium-230 is expected to have a Kd value similar to native thorium. The distribution coefficient of thorium is highly variable, particularly humic substances are known to influence the sorption. The estimated Kd value of $3.9x10^{-3}$ m^3/g for thorium-230 is near the lower limit of the range. The sorption coefficient is consistent with the calculated and measured Kd values reported by Kaplan and Serkiz (2000), and is expected for the low organic environment of the Judge Sissons Lake sediments.

The estimated Kd value for radium-226 is similar to that for thorium-230. The Kd values are consistent with the reported distribution of thorium-230 and radium-226 at various locations in northern Saskatchewan (Neame et al, 1988).

The metalloid polonium resembles bismuth and tellurium in chemical properties. The somewhat lower Kd value of polonium-210 ($1.1x10^{-3}$ m^3/g) is likely due to its semi-metal character.

Another dataset of site-specific Kd values, developed from solid phase concentrations and surface water concentrations, was used to characterize the variance. The site-specific Kd_{sed} values were assumed to represent the average and the standard deviation from the Kd values was used to estimate a geometric mean for the lognormal distribution. Similarly, the geometric standard deviation calculated from the Kd values was used as the geometric standard deviation for the lognormal distribution. These distributions were used as a starting point and were further calibrated based on the model results and sediment behaviour under baseline conditions. Table 5.1-2 provides the calibrated site-specific Kd_{sed} values used for the assessment. It is noted that Kd_{sed} values are not presented for the conventional constituents. Ammonia, calcium, chloride, sulphate and TDS were assumed to not undergo removal from the water column to sediment through sorption; therefore, water-to-sediment distribution coefficients were not necessary for these parameters.

Table 5.1-2 Summary of Site-Specific Water-to-Sediment Distribution Coefficients – Kd_{sed}

LogNormal Distribution for Water-to-Sediment Distribution Coefficients Kd_{sed} (m^3/(g dw))				
Constituent	**Geometric Mean**	**GSD**	**Min**	**Max**
Uranium	$7.5x10^{-3}$	2.0	$9.4x10^{-4}$	$6.0x10^{-2}$
Thorium-230	$7.5x10^{-4}$	2.0	$9.4x10^{-5}$	$6.0x10^{-3}$
Lead-210	$6.7x10^{-3}$	1.4	$2.4x10^{-3}$	$1.8x10^{-2}$
Radium-226	$2.1x10^{-3}$	1.5	$6.2x10^{-4}$	$7.1x10^{-3}$
Polonium-210	$6.7x10^{-3}$	1.4	$2.4x10^{-3}$	$1.8x10^{-2}$
Arsenic	$5.5x10^{-3}$	2.0	$6.9x10^{-4}$	$4.4x10^{-2}$
Cadmium	$3.0x10^{-4}$	1.5	$8.9x10^{-5}$	$1.0x10^{-3}$
Cobalt	$4.1x10^{-2}$	1.5	$1.2x10^{-2}$	$1.4x10^{-1}$
Copper	$4.1x10^{-3}$	1.5	$1.2x10^{-3}$	$1.4x10^{-2}$
Lead	$1.5x10^{-2}$	2.0	$1.9x10^{-3}$	$1.2x10^{-1}$
Molybdenum	$1.5x10^{-3}$	2.0	$1.9x10^{-4}$	$1.2x10^{-2}$
Nickel	$4.6x10^{-3}$	2.0	$5.7x10^{-4}$	$3.6x10^{-2}$
Selenium	$1.0x10^{-3}$	1.6	$2.3x10^{-4}$	$4.4x10^{-3}$
Zinc	$6.3x10^{-3}$	2.0	$7.9x10^{-4}$	$5.0x10^{-2}$

Note: calibrated site-specific Kdsed values
Lognormal distribution (Geomean, Geo Std Dev, Min, Max).

µg/g = micrograms per gram; Bq/g = Bequerels per gram.
dw = dry weight basis; ww = wet weight basis.

5.1.2 Aquatic Biota Transfer Factors

Section 2.8 provided the aquatic biota model equations for the use of transfer factors. The following sections present the transfer factors used in the current assessment and basis for their development.

5.1.2.1 Fish Transfer Factors

The fish chemistry database from work carried out in 2002 and 2005 at both the McClean Lake and Rabbit Lake sites in northern Saskatchewan were pooled and analyzed to investigate alternative methods for estimating fish concentrations based on measured water concentrations. A minimum level in fish approach is suggested, which is set based on analysis of data from areas where the water concentrations was at or near the method detection limit (MDL). A linear transfer factor is used to estimate fish concentrations. These transfer factors were estimated from the data collected in areas with higher concentrations in the water. This approach is similar in concept to the "hockey-stick regression" in the Toll-Brix model (Toll et al. 2005, Brix et al. 2005). The site-specific data for water and fish collected from the site were analyzed with transfer factors developed from other sites and it was determined that the Kiggavik fish data was consistent with the method outlined above. The

parameters of the fish model are outlined inTable 5.1-3. Attachment E provides a comparison of the model calculated fish concentrations with measured fish concentrations from the site.

Table 5.1-3 Summary of Fish Concentration Model

Constituent	Geometric Mean	GSD	Min	Max
LogNormal Distribution for Minimum Fish Concentrations (µg/g ww or Bq/g ww)				
Uranium	$1.1x10^{-3}$	2.8	$5.3x10^{-5}$	$2.4x10^{-2}$
Uranium-238	$2.6x10^{-5}$	2.8	$1.3x10^{-6}$	$5.8x10^{-4}$
Thorium-230	$1.1x10^{-4}$	2.5	$7.1x10^{-6}$	$1.6x10^{-3}$
Lead-210	$6.2x10^{-4}$	1.9	$8.8x10^{-5}$	$4.3x10^{-3}$
Radium-226	$7.0x10^{-5}$	2.3	$5.5x10^{-6}$	$8.8x10^{-4}$
Polonium-210	$4.6x10^{-4}$	3.0	$1.7x10^{-5}$	$1.3x10^{-2}$
Arsenic	$4.9x10^{-2}$	2.4	$3.4x10^{-3}$	$7.2x10^{-1}$
Cadmium	$6.2x10^{-3}$	4.7	$6.1x10^{-5}$	$6.3x10^{-1}$
Cobalt	$4.8x10^{-3}$	1.7	$9.9x10^{-4}$	$2.3x10^{-2}$
Copper	$2.3x10^{-1}$	1.6	$5.6x10^{-2}$	$9.5x10^{-1}$
Lead	$1.3x10^{-2}$	4.0	$2.0x10^{-4}$	$8.3x10^{-1}$
Molybdenum	$1.8x10^{-2}$	1.7	$3.4x10^{-3}$	$9.3x10^{-2}$
Nickel	$1.2x10^{-2}$	2.0	$1.4x10^{-3}$	$9.5x10^{-2}$
Selenium	$3.7x10^{-1}$	1.5	$1.0x10^{-1}$	1.4
Zinc	5.1	1.5	1.5	17.
LogNormal Distribution for Water-to-Fish Transfer Factors (m^3/g ww)				
Uranium	$4.8x10^{-7}$	2.0	$5.9x10^{-8}$	$4.0x10^{-6}$
Thorium-230	$3.6x10^{-5}$	3.0	$6.9x10^{-6}$	$1.9x10^{-4}$
Lead-210	$8.5x10^{-5}$	1.1	$6.6x10^{-5}$	$1.1x10^{-4}$
Radium-226	$9.3x10^{-5}$	1.6	$2.5x10^{-5}$	$3.5x10^{-4}$
Polonium-210	$2.5x10^{-4}$	2.7	$1.3x10^{-5}$	$5.1x10^{-3}$
Arsenic	$2.4x10^{-5}$	1.5	$7.4x10^{-6}$	$7.5x10^{-5}$
Cadmium	$2.0x10^{-4}$	2.5	$5.1x10^{-5}$	$7.9x10^{-4}$
Cobalt	$5.7x10^{-5}$	1.7	$1.2x10^{-5}$	$2.8x10^{-4}$
Copper	$3.7x10^{-5}$	1.9	$1.5x10^{-5}$	$9.3x10^{-5}$
Lead	$2.2x10^{-5}$	4.0	$3.5x10^{-7}$	$1.3x10^{-3}$
Molybdenum	$1.1x10^{-7}$	5.2	$7.6x10^{-10}$	$1.6x10^{-6}$
Nickel	$1.6x10^{-6}$	2.0	$2.0x10^{-7}$	$1.2x10^{-5}$
Selenium	$2.2x10^{-3}$	1.5	$1.1x10^{-3}$	$4.1x10^{-3}$
Zinc	$1.2x10^{-3}$	1.6	$3.2x10^{-4}$	$4.7x10^{-3}$

Table 5.1-3 Summary of Fish Concentration Model

Constituent	Geometric Mean	GSD	Min	Max
Note: Fish model parameters: minimum fish concentrations represent fish concentrations at the lowest water concentrations; transfer factors dominate once the water concentration results in a fish concentration above the minimum fish concentration. Lognormal distribution (Geometric Mean, GSD, Min, Max). m3/g = cubic meters per gram. ww = wet weight basis.				

As mentioned above, the transfer factors used for fish in the assessment are based on data from the McClean, Rabbit Lake, and Kiggavik sites. The fish species typically evaluated at these sites include northern pike, lake trout, lake whitefish, and white sucker with Arctic grayling and cisco species also collected from the Kiggavik site. The use of measured site data to develop transfer factors inherently considers any trophic differences or biomagnification that occurs in different species or trophic levels.

5.1.2.2 Aquatic Vegetation Transfer Factors

Site-specific water-to-aquatic vegetation transfer factors were developed using the baseline water quality (Section 3.2.3) and baseline aquatic vegetation quality (Section 3.2.6). Lognormal distributions, summarized in Table 5.1-4, were derived from the site-specific data. Attachment E provides a comparison of the model calculated aquatic vegetation concentrations with measured aquatic vegetation concentrations from the site.

Table 5.1-4 Summary of Site-Specific Transfer Factors – Aquatic Vegetation

Constituent	Geometric Mean	GSD	Min	Max
LogNormal Distribution for Water-to-Aquatic Vegetation Transfer Factor (m^3/g ww)				
Uranium	$5.5x10^{-5}$	3.6	$1.2x10^{-6}$	$2.6x10^{-3}$
Thorium-230[a]	$4.6x10^{-4}$	2.4	$3.3x10^{-5}$	$6.4x10^{-3}$
Lead-210	$6.0x10^{-4}$	1.9	$9.3x10^{-5}$	$3.9x10^{-3}$
Radium-226	$1.2x10^{-4}$	2.1	$1.2x10^{-5}$	$1.2x10^{-3}$
Polonium-210	$1.6x10^{-3}$	2.1	$1.6x10^{-4}$	$1.6x10^{-2}$
Arsenic	$1.0x10^{-3}$	8.5	$1.7x10^{-6}$	$6.3x10^{-1}$
Cadmium[b]	$4.2x10^{-4}$	2.5	$2.7x10^{-5}$	$6.5x10^{-3}$
Cobalt	$8.0x10^{-4}$	4.9	$6.8x10^{-6}$	$9.5x10^{-2}$
Copper	$1.6x10^{-3}$	2.0	$2.0x10^{-4}$	$1.3x10^{-2}$
Lead	$5.0x10^{-4}$	6.3	$2.0x10^{-6}$	$1.2x10^{-1}$
Molybdenum	$1.1x10^{-3}$	2.6	$5.8x10^{-5}$	$2.0x10^{-2}$
Nickel	$5.7x10^{-4}$	2.6	$3.4x10^{-5}$	$9.6x10^{-3}$

Table 5.1-4 Summary of Site-Specific Transfer Factors – Aquatic Vegetation

Constituent	Geometric Mean	GSD	Min	Max
LogNormal Distribution for Water-to-Aquatic Vegetation Transfer Factor (m^3/g ww)				
Selenium	$1.4x10^{-4}$	2.0	$1.8x10^{-5}$	$1.1x10^{-3}$
Zinc	$1.0x10^{-3}$	2.4	$7.0x10^{-5}$	$1.4x10^{-2}$
Note:site-specific TFwater-to-aqveg values a – calibrated from SLEEA (SENES 2008, based on SENES 2006), not from site-specific data. b – calibrated from Bird and Schwartz (1996). Lognormal distribution (Geometric Mean, GSD, Min, Max). m3/g = cubic meters per gram. ww = wet weight basis.				

5.1.2.3 Benthic Invertebrate Transfer Factors

Water-to-benthic invertebrate transfer factors were taken from site-specific data for the McClean Lake site in northern Saskatchewan collected in Summer 2010. Hard- and soft-bodied benthic invertebrate samples were collected from eight locations. These data were consolidated and represented by lognormal distributions, with the geometric mean calculated from the data and the GSD derived to represent the arithmetic mean of the data. The minimum and maximum values were defined at 3 standard deviations. Data from the McClean site was used in favour of literature data, since it was considered to be more relevant to the Kiggavik site. Model predicted concentrations for benthic invertebrates were compared with measured benthic data collected from the site in 2013, and transfer factors for select COPC were modified to improve the comparison. The distributions are summarized in Table 5.1-5. Attachment E provides a comparison of the model calculated benthic invertebrate concentrations with measured insect and benthic invertebrate concentrations from the site.

Table 5.1-5 Summary of Transfer Factors – Benthic Invertebrates

Constituent	Geometric Mean	GSD	Min	Max
LogNormal Distribution for Water-to-Benthic Invertebrate Transfer Factor (m^3/g ww)				
Uranium	7.1x10-4	1.5	2.3x10-4	2.2x10-3
Thorium-230	6.0x10-5	2.5	3.8x10-6	9.3x10-4
Lead-210	2.2x10-4	2.5	1.4x10-5	3.4x10-3
Radium-226	9.7x10-4	2.5	6.2x10-5	1.5x10-2
Polonium-210	1.5x10-3	2.5	9.7x10-5	2.4x10-2
Arsenica	2.7x10-3	2.2	2.5x10-4	2.8x10-2
Cadmiumb	1.4x10-3	2.5	6.5x10-5	7.4x10-3
Cobalta	4.1x10-3	2.6	2.3x10-4	7.0x10-2
Coppera	1.4x10-2	1.5	4.2x10-3	4.8x10-2

Table 5.1-5 Summary of Transfer Factors – Benthic Invertebrates

Constituent	Geometric Mean	GSD	Min	Max
LogNormal Distribution for Water-to-Benthic Invertebrate Transfer Factor (m^3/g ww)				
Leada	1.3x10-3	2.9	5.4x10-5	3.2x10-2
Molybdenuma	9.5x10-4	2.2	8.9x10-5	1.0x10-2
Nickela	8.1x10-4	3.6	1.7x10-5	3.8x10-2
Selenium	6.2x10-4	2.5	4.0x10-5	9.8x10-3
Zinc	5.7x10-3	2.9	2.3x10-4	1.4x10-1
Note: TFwater-to-benthic values from McClean Lake site data a – calibrated based on 2013 site data. b – from U.S. EPA (1979) and COGEMA (1997). Lognormal distribution (Geometric Mean, GSD, Min, Max). m3/g = cubic meters per gram. ww = wet weight basis.				

5.1.2.4 Phytoplankton and Zooplankton Transfer Factors

Water-to-phytoplankton and water-to-zooplankton transfer factors are required for the aquatic radiological assessment to evaluate the internal dose to these aquatic receptors. Transfer factors were taken from literature values and are summarized in Table 5.1-6.

Table 5.1-6 Summary of Transfer Factors – Phytoplankton and Zooplankton

Constituent	Phytoplankton		Zooplankton	
	Water-to-Phytoplankton Transfer Factor (m^3/g ww)	Reference	Water-to-Zooplankton Transfer Factor (m3/g ww)	Reference
Uranium	$1.20x10^{-4}$	Hosseini et al. 2008	$1.70x10^{-4}$	IAEA 2009
Thorium-230	$4.00x10^{-3}$	Hosseini et al. 2008	$2.90x10^{-3}$	IAEA 2009
Lead-210	$4.90x10^{-1}$	Hosseini et al. 2008	$2.50x10^{-5}$	IAEA DRAFT
Radium-226	$3.30x10^{-4}$	IAEA DRAFT	$1.40x10^{-4}$	IAEA DRAFT
Polonium-210	$2.70x10^{-2}$	Hosseini et al. 2008	$6.30x10^{-3}$	IAEA DRAFT

5.1.3 Terrestrial Biota Transfer Factors

Section 2.9 provided the terrestrial biota model equations for the use of transfer factors. The following sections present the transfer factors used in the current assessment and basis for their development.

5.1.3.1 Terrestrial Vegetation Transfer Factors

The measured data for terrestrial vegetation at the site was used with the baseline soil data to develop site-specific transfer factors for terrestrial vegetation. The transfer factors are used to predict concentrations in terrestrial vegetation in the future to reflect potential changes in soil concentrations.

Table 5.1-6 summarizes the site-specific transfer factors developed for browse, forage, and berry. Attachment E provides a comparison of the model calculated terrestrial vegetation concentrations with measured terrestrial vegetation concentrations from the site.

Table 5.1-7 Summary of Site-Specific Transfer Factors – Terrestrial Vegetation

Constituent	Geometric Mean	GSD	Min	Max
LogNormal Distribution for Soil-to-Browse Transfer Factor ((Bq/g ww)/(µg/g dw) and (µg/g ww)/(µg/g dw))				
Uranium	0.0009	2.0[a]	0.0001	0.007
Thorium-230[b]	0.009	1.7	0.002	0.04
Lead-210[c]	1.7	1.5	0.49	6.2
Radium-226	0.04	1.8	0.008	0.22
Polonium-210	1.7	1.5	0.49	6.2
Arsenic	0.027	3.2[d]	0.0008	0.87
Cadmium[e]	1.0	2.5	0.064	15.6
Cobalt	0.06	3.1	0.002	1.9
Copper	0.63	1.8	0.11	3.5
Lead	0.013	1.9	0.002	0.09
Molybdenum	0.12	4.5	0.001	10.9
Nickel	0.1	2.1	0.01	0.92
Selenium[b]	0.01	3.2	0.001	0.1
Zinc	8.8	2.0[a]	2.2	35.2
LogNormal Distribution for Soil-to-Forage Transfer Factor ((Bq/g ww)/(µg/g dw) and (µg/g ww)/(µg/g dw))				
Uranium	0.04	3.0[a]	0.0015	1.1
Thorium-230	0.09	4.0[a]	0.0002	42.5
Lead-210	3.0	3.6[d]	0.07	136
Radium-226	0.06	7.8	0.0001	29.6
Polonium-210	2.2	2.8[a]	0.1	48.3
Arsenic[f]	0.005	4.5	0.00005	0.46
Cadmium[g]	0.2	2.5	0.032	1.3
Cobalt	0.03	3.0[a]	0.001	0.81
Copper	0.26	2.0[a]	0.033	2.1
Lead	0.02	2.5[a]	0.001	0.31

Table 5.1-7 Summary of Site-Specific Transfer Factors – Terrestrial Vegetation

Constituent	Geometric Mean	GSD	Min	Max
Molybdenum	0.24	3.0[a]	0.009	6.5
Nickel	0.07	2.5[a]	0.004	1.1
Selenium[b]	Uniform Distribution		0.004	0.5
Zinc	1.1	2.0[a]	0.14	8.8
LogNormal Distribution for Soil-to-Berry Transfer Factor ((Bq/g ww)/(µg/g dw) and (µg/g ww)/(µg/g dw))				
Uranium[b,f]	0.0011	2.0	0.00014	0.0088
Thorium-230	0.002	13[d]	0.000001	4.9
Lead-210[b]	0.013	5.2	0.007	0.13
Radium-226	0.004	2.9	0.0002	0.1
Polonium-210	0.02	1.9	0.003	0.15
Arsenic	0.004	2.5[d]	0.0003	0.067
Cadmium[e]	0.05	2.0	0.0063	0.4
Cobalt	0.009	1.8	0.002	0.05
Copper	0.11	2.4	0.008	1.6
Lead	0.002	2.5	0.0002	0.04
Molybdenum[f]	0.018	4.0	0.00028	1.2
Nickel	0.03	2.6	0.002	0.5
Selenium[b]	0.0029	2.5	0.0002	0.045
Zinc	0.65	3.3	0.02	23.3

Note: Lognormal distribution (Geomean, Geo Std Dev, Min, Max)
a – calibrated GSD
b – distribution from SLEEA
c – assumed same as Po-210
d – GSD from SLEEA
e – from NCRP (1996)
f – calibrated
g – from NCRP(1996), EPA (1998)
µg/g = micrograms per gram; Bq/g = Bequerels per gram.
dw = dry weight basis; ww = wet weight basis.

As described in Section 2.9.3, lichen concentrations are estimated in the model using a COPC-specific component to represent uptake, retention, and subsequent release. This COPC-specific term was calibrated using measured lichen data from the site and verified using lichen data from other sites in northern Saskatchewan. Attachment E provides a comparison of the model calculated lichen concentrations with measured lichen concentrations from the site.

Table 5.1-8 Summary of Site-Specific Transfer Factors – Lichen

Constituent	Min	Max
Uniform Distribution for COPC-Specific Lichen Transfer Factor ((Bq/g ww)/(µg/g dw) and (µg/g ww)/(µg/g dw))		
Uranium	$7.8x10^{-10}$	$3.1x10^{-9}$
Thorium-230	$4.7x10^{-9}$	$1.9x10^{-8}$
Lead-210	$7.5x10^{-10}$	$3.0x10^{-9}$
Radium-226	$1.9x10^{-9}$	$7.6x10^{-9}$
Polonium-210	$1.8x10^{-10}$	$7.1x10^{-10}$
Arsenic	$9.8x10^{-10}$	$3.9x10^{-9}$
Cadmium[e]	$4.4x10^{-10}$	$1.8x10^{-9}$
Cobalt	$1.5x10^{-10}$	$5.8x10^{-10}$
Copper	$1.3x10^{-8}$	$5.0x10^{-8}$
Lead	$5.1x10^{-9}$	$2.1x10^{-8}$
Molybdenum	$5.5x10^{-9}$	$2.2x10^{-8}$
Nickel	$2.6x10^{-10}$	$1.0x10^{-9}$
Selenium[b]	$3.1x10^{-10}$	$1.2x10^{-9}$
Zinc	$4.0x10^{-10}$	$1.6x10^{-9}$

5.1.3.2 Bird Transfer Factors

Feed-to-bird transfer factors are used in the model to estimate the concentration of COPC in the flesh of birds, based on predicted intake rates of COPC, as shown Section 2.9.

Transfer factors are generally derived for domestic agricultural animals (cattle and poultry). For chickens, according to the Chicken Farmers of Canada (CFC), chickens are ready for market at approximately 2 kg (CFC 2007, internet site). The typical body weight at 42 days is 2.7 kg (Havenstein *et al.* 2003); this value was used in the calculations.

There is limited data available on avian species. Clulow et al. (1992) measured a concentration ratio of 0.075 on a fresh weight basis for grouse. Using the appropriate intake rate a transfer factor of 0.3 d/kg can be determined. Using the poultry value (0.13 d/kg, Sheppard et al., 2005), the TF for radium exposure to a grouse is estimated to be 0.4 d/kg.

Due to the lack of available transfer factors for different non-human biota for all elements, it is recommended that allometric scaling of the transfer factors based on beef or chicken is used. This approach provides a reasonable estimate of the transfer factor that can be used to calculate the concentration of non-human biota.

To obtain a more appropriate transfer factor, allometric scaling can be applied to the transfer factor with a relationship of -0.75. This approach is consistent with the allometric scaling for intake rates and inhalation by wildlife, as used in the ecological profiles (U.S. EPA 1993), which has shown a similar relationship. Allometric scaling of transfer factors has been discussed by others (e.g., Nalezinski *et al.* 1996; Higley *et al.* 2003) as a useful method for deriving transfer factors for biota. It is acknowledged that not all radionuclides would scale to the same factor, as shown by the U.S. DOE (2002). However, the use of the -0.75 factor is a conservative approach. Other factors that can be found in the literature (e.g., 0.25 may be appropriate for actinides) would result in smaller predicted transfer factors for smaller biota than the reference animal. As most of the ecological receptors are smaller than cattle, the -0.75 is used as a conservative approach.

The scaling can be applied as follows in equation (5.1-1):

$$TF_w = TF_A \left(\frac{BW_w}{BW_A} \right)^{-0.75} \qquad (5.1\text{-}1)$$

where:

TF_w	Transfer factor for wildlife (d/kg)
TF_A	Transfer factor for animal available from literature (d/kg)
BW_w	Body weight of wildlife (kg)
BW_A	Body weight of animal (kg)

Scaling for the selected bird species was conducted from a selected reference bird species.

When the allometric equations for the transfer factor are combined with the allometric equation for food intake, it results in generally a constant trans-species ratio between whole-body and dietary activity concentrations (e.g., Beresford 2007). This approach was used in the Framework for Assessment of Environmental Impact (FASSET 2003). This approach would be appropriate however, as data in literature is generally reported as transfer factors, as opposed to concentration ratios, the use of the selected approach allowed a larger database to be selected.

Feed-to-bird transfer factors from literature were used as a starting point and model outputs for predicted concentrations in Lapland longspur under baseline conditions were compared with the measured site data for sparrow (Table 3.2-12). This comparison was considered reasonable because Lapland longspur is described as being “sparrow-like” and has a similar diet and body weight to the sparrow. Modifications were made to the feed-to-bird transfer factors from literature to achieve comparable concentrations in the Lapland longspur. Feed-to-bird transfer factors were scaled by body weight for each avian VEC using equation (5.1-2).

$$TF_{VEC} = TF_{bird} \times \left(\frac{BW_{VEC}}{BW_{bird}} \right)^{-0.75} \qquad (5.1\text{-}2)$$

Table 5.1-9 presents the calibrated feed-to-bird transfer factors used in the assessment. Lognormal probabilistic distributions were derived using an assumed geometric standard deviation of 2.5. Attachment E provides a comparison of the model calculated bird concentrations with measured bird concentrations from the site.

Table 5.1-9 Summary of Feed-to-Bird Transfer Factors

Constituent	Geometric Mean	GSD	Min	Max
LogNormal Distribution for Feed-to-Falcon Transfer Factors (d/g ww)				
Uranium	$1.3x10^{-4}$	2.5	$8.6x10^{-6}$	$2.1x10^{-3}$
Thorium-230	$6.0x10^{-4}$	2.5	$3.8x10^{-5}$	$9.4x10^{-3}$
Lead-210	$1.5x10^{-2}$	2.5	$9.5x10^{-4}$	$2.3x10^{-1}$
Radium-226	$1.2x10^{-3}$	2.5	$7.8x10^{-5}$	$1.9x10^{-2}$
Polonium-210	$4.6x10^{-3}$	2.5	$2.9x10^{-4}$	$7.2x10^{-2}$
Arsenic	$1.5x10^{-4}$	2.5	$9.7x10^{-6}$	$2.4x10^{-3}$
Cadmium	$3.2x10^{-4}$	2.5	$2.1x10^{-5}$	$5.1x10^{-3}$
Cobalt	$3.6x10^{-4}$	2.5	$2.3x10^{-5}$	$5.6x10^{-3}$
Copper	$1.1x10^{-3}$	2.5	$6.9x10^{-5}$	$1.7x10^{-2}$
Lead	$4.0x10^{-4}$	2.5	$2.5x10^{-5}$	$6.2x10^{-3}$
Molybdenum	$1.2x10^{-3}$	2.5	$7.5x10^{-5}$	$1.8x10^{-2}$
Nickel	$1.8x10^{-4}$	2.5	$1.2x10^{-5}$	$2.8x10^{-3}$
Selenium	$9.3x10^{-3}$	2.5	$6.0x10^{-4}$	$1.5x10^{-1}$
Zinc	$1.0x10^{-3}$	2.5	$6.7x10^{-5}$	$1.6x10^{-2}$
LogNormal Distribution for Feed-to-Longspur Transfer Factors (d/g ww)				
Uranium	$1.7x10^{-3}$	2.5	$1.1x10^{-4}$	$2.7x10^{-2}$
Thorium-230	$7.7x10^{-3}$	2.5	$4.9x10^{-4}$	$1.2x10^{-1}$
Lead-210	$1.9x10^{-1}$	2.5	$1.2x10^{-2}$	3.0
Radium-226	$1.6x10^{-2}$	2.5	$1.0x10^{-3}$	$2.4x10^{-1}$
Polonium-210	$5.9x10^{-2}$	2.5	$3.8x10^{-3}$	$9.2x10^{-1}$
Arsenic	$1.9x10^{-3}$	2.5	$1.2x10^{-4}$	$3.0x10^{-2}$
Cadmium	$4.2x10^{-3}$	2.5	$2.7x10^{-4}$	$6.5x10^{-2}$
Cobalt	$4.6x10^{-3}$	2.5	$2.9x10^{-4}$	$7.2x10^{-2}$
Copper	$1.4x10^{-2}$	2.5	$8.8x10^{-4}$	$2.1x10^{-1}$
Lead	$5.1x10^{-3}$	2.5	$3.3x10^{-4}$	$8.0x10^{-2}$
Molybdenum	$1.5x10^{-2}$	2.5	$9.6x10^{-4}$	$2.3x10^{-1}$
Nickel	$2.3x10^{-3}$	2.5	$1.5x10^{-4}$	$3.6x10^{-2}$
Selenium	$1.2x10^{-1}$	2.5	$7.7x10^{-3}$	1.9
Zinc	$1.3x10^{-2}$	2.5	$8.6x10^{-4}$	$2.1x10^{-1}$
LogNormal Distribution for Feed-to-Longtail Duck Transfer Factors (d/g ww)				
Uranium	$1.5x10^{-4}$	2.5	$9.4x10^{-6}$	$2.3x10^{-3}$

Table 5.1-9 Summary of Feed-to-Bird Transfer Factors

Constituent	Geometric Mean	GSD	Min	Max
Thorium-230	$6.5x10^{-4}$	2.5	$4.2x10^{-5}$	$1.0x10^{-2}$
Lead-210	$1.6x10^{-2}$	2.5	$1.0x10^{-3}$	$2.5x10^{-1}$
Radium-226	$1.3x10^{-3}$	2.5	$8.5x10^{-5}$	$2.1x10^{-2}$
Polonium-210	$5.0x10^{-3}$	2.5	$3.2x10^{-4}$	$7.9x10^{-2}$
Arsenic	$1.7x10^{-4}$	2.5	$1.1x10^{-5}$	$2.6x10^{-3}$
Cadmium	$3.5x10^{-4}$	2.5	$2.3x10^{-5}$	$5.5x10^{-3}$
Cobalt	$3.9x10^{-4}$	2.5	$2.5x10^{-5}$	$6.1x10^{-3}$
Copper	$1.2x10^{-3}$	2.5	$7.5x10^{-5}$	$1.8x10^{-2}$
Lead	$4.3x10^{-4}$	2.5	$2.8x10^{-5}$	$6.8x10^{-3}$
Molybdenum	$1.3x10^{-3}$	2.5	$8.1x10^{-5}$	$2.0x10^{-2}$
Nickel	$2.0x10^{-4}$	2.5	$1.3x10^{-5}$	$3.1x10^{-3}$
Selenium	$1.0x10^{-2}$	2.5	$6.5x10^{-4}$	$1.6x10^{-1}$
Zinc	$1.1x10^{-3}$	2.5	$7.3x10^{-5}$	$1.8x10^{-2}$
LogNormal Distribution for Feed-to-Merganser Transfer Factors (d/g ww)				
Uranium	$1.2x10^{-4}$	2.5	$8.0x10^{-6}$	$1.9x10^{-3}$
Thorium-230	$5.5x10^{-4}$	2.5	$3.5x10^{-5}$	$8.6x10^{-3}$
Lead-210	$1.4x10^{-2}$	2.5	$8.8x10^{-4}$	$2.1x10^{-1}$
Radium-226	$1.1x10^{-3}$	2.5	$7.2x10^{-5}$	$1.8x10^{-2}$
Polonium-210	$4.3x10^{-3}$	2.5	$2.7x10^{-4}$	$6.6x10^{-2}$
Arsenic	$1.4x10^{-4}$	2.5	$9.0x10^{-6}$	$2.2x10^{-3}$
Cadmium	$3.0x10^{-4}$	2.5	$1.9x10^{-5}$	$4.7x10^{-3}$
Cobalt	$3.3x10^{-4}$	2.5	$2.1x10^{-5}$	$5.2x10^{-3}$
Copper	$9.9x10^{-4}$	2.5	$6.3x10^{-5}$	$1.5x10^{-2}$
Lead	$3.7x10^{-4}$	2.5	$2.4x10^{-5}$	$5.7x10^{-3}$
Molybdenum	$1.1x10^{-3}$	2.5	$6.9x10^{-5}$	$1.7x10^{-2}$
Nickel	$1.7x10^{-4}$	2.5	$1.1x10^{-5}$	$2.6x10^{-3}$
Selenium	$8.6x10^{-3}$	2.5	$5.5x10^{-4}$	$1.3x10^{-1}$
Zinc	$9.7x10^{-4}$	2.5	$6.2x10^{-5}$	$1.5x10^{-2}$
LogNormal Distribution for Feed-to-Pintail Transfer Factors (d/g ww)				
Uranium	$1.2x10^{-4}$	2.5	$8.0x10^{-6}$	$1.9x10^{-3}$
Thorium-230	$5.5x10^{-4}$	2.5	$3.5x10^{-5}$	$8.6x10^{-3}$
Lead-210	$1.4x10^{-2}$	2.5	$8.8x10^{-4}$	$2.1x10^{-1}$
Radium-226	$1.1x10^{-3}$	2.5	$7.2x10^{-5}$	$1.8x10^{-2}$
Polonium-210	$4.3x10^{-3}$	2.5	$2.7x10^{-4}$	$6.6x10^{-2}$
Arsenic	$1.4x10^{-4}$	2.5	$9.0x10^{-6}$	$2.2x10^{-3}$
Cadmium	$3.0x10^{-4}$	2.5	$1.9x10^{-5}$	$4.7x10^{-3}$
Cobalt	$3.3x10^{-4}$	2.5	$2.1x10^{-5}$	$5.2x10^{-3}$

Table 5.1-9 Summary of Feed-to-Bird Transfer Factors

Constituent	Geometric Mean	GSD	Min	Max
Copper	$9.9x10^{-4}$	2.5	$6.3x10^{-5}$	$1.5x10^{-2}$
Lead	$3.7x10^{-4}$	2.5	$2.4x10^{-5}$	$5.7x10^{-3}$
Molybdenum	$1.1x10^{-3}$	2.5	$6.9x10^{-5}$	$1.7x10^{-2}$
Nickel	$1.7x10^{-4}$	2.5	$1.1x10^{-5}$	$2.6x10^{-3}$
Selenium	$8.6x10^{-3}$	2.5	$5.5x10^{-4}$	$1.3x10^{-1}$
Zinc	$9.7x10^{-4}$	2.5	$6.2x10^{-5}$	$1.5x10^{-2}$
LogNormal Distribution for Feed-to-Ptarmigan Transfer Factors (d/g ww)				
Uranium	$1.8x10^{-4}$	2.5	$1.1x10^{-5}$	$2.8x10^{-3}$
Thorium-230	$7.9x10^{-4}$	2.5	$5.1x10^{-5}$	$1.2x10^{-2}$
Lead-210	$2.0x10^{-2}$	2.5	$1.3x10^{-3}$	$3.1x10^{-1}$
Radium-226	$1.6x10^{-3}$	2.5	$1.0x10^{-4}$	$2.5x10^{-2}$
Polonium-210	$6.1x10^{-3}$	2.5	$3.9x10^{-4}$	$9.5x10^{-2}$
Arsenic	$2.0x10^{-4}$	2.5	$1.3x10^{-5}$	$3.1x10^{-3}$
Cadmium	$4.3x10^{-4}$	2.5	$2.7x10^{-5}$	$6.7x10^{-3}$
Cobalt	$4.8x10^{-4}$	2.5	$3.0x10^{-5}$	$7.4x10^{-3}$
Copper	$1.4x10^{-3}$	2.5	$9.1x10^{-5}$	$2.2x10^{-2}$
Lead	$5.3x10^{-4}$	2.5	$3.4x10^{-5}$	$8.2x10^{-3}$
Molybdenum	$1.5x10^{-3}$	2.5	$9.9x10^{-5}$	$2.4x10^{-2}$
Nickel	$2.4x10^{-4}$	2.5	$1.5x10^{-5}$	$3.8x10^{-3}$
Selenium	$1.2x10^{-2}$	2.5	$7.9x10^{-4}$	$1.9x10^{-1}$
Zinc	$1.4x10^{-3}$	2.5	$8.9x10^{-5}$	$2.2x10^{-2}$
LogNormal Distribution for Feed-to-Sandpiper Transfer Factors (d/g ww)				
Uranium	$1.7x10^{-3}$	2.5	$1.1x10^{-4}$	$2.7x10^{-2}$
Thorium-230	$7.7x10^{-3}$	2.5	$4.9x10^{-4}$	$1.2x10^{-1}$
Lead-210	$1.9x10^{-1}$	2.5	$1.2x10^{-2}$	3.0
Radium-226	$1.6x10^{-2}$	2.5	$1.0x10^{-3}$	$2.4x10^{-1}$
Polonium-210	$5.9x10^{-2}$	2.5	$3.8x10^{-3}$	$9.2x10^{-1}$
Arsenic	$1.9x10^{-3}$	2.5	$1.2x10^{-4}$	$3.0x10^{-2}$
Cadmium	$4.2x10^{-3}$	2.5	$2.7x10^{-4}$	$6.5x10^{-2}$
Cobalt	$4.6x10^{-3}$	2.5	$2.9x10^{-4}$	$7.2x10^{-2}$
Copper	$1.4x10^{-2}$	2.5	$8.8x10^{-4}$	$2.1x10^{-1}$
Lead	$5.1x10^{-3}$	2.5	$3.3x10^{-4}$	$8.0x10^{-2}$
Molybdenum	$1.5x10^{-2}$	2.5	$9.6x10^{-4}$	$2.3x10^{-1}$
Nickel	$2.3x10^{-3}$	2.5	$1.5x10^{-4}$	$3.6x10^{-2}$
Selenium	$1.2x10^{-1}$	2.5	$7.7x10^{-3}$	1.9
Zinc	$1.3x10^{-2}$	2.5	$8.6x10^{-4}$	$2.1x10^{-1}$

Source: Literature values calibrated to measured sparrow site data.
Note: GSD = geometric standard deviation.

Table 5.1-9 Summary of Feed-to-Bird Transfer Factors

Constituent	Geometric Mean	GSD	Min	Max

Min = minimum; Max = maximum.

d/g = day per gram.
ww = wet weight basis.

5.1.3.3 Animal Transfer Factors

Feed-to-animal transfer factors are used in the model to estimate the concentration of COPC in the flesh of animals, based on predicted intake rates of COPC, as shown Section 2.9.

Allometric scaling for the feed-to-animal transfer factors was completed similar to the allometric scaling described in the previous section for feed-to-bird transfer factors. The scaling for the selected mammalian species was conducted from a reference mammal species.

Transfer factors are generally derived for domestic agricultural animals (cattle and poultry). Beef cattle were assumed to be the agricultural animal of interest and thus the body weight can be taken to be approximately 400 kg (NRC 2000; CCA 1999, internet site).

Note that this approach provides transfer factors that are consistent with the limited amount of measured values that are available. This can be illustrated using the transfer factor for lead. A feed to beef transfer 0.0004 d/kg can be obtained from literature (IAEA 1994). Using a body weight of 400 kg for cattle and 0.03 kg for a vole allometric scaling can be applied. This provides a predicted transfer factor for a vole of 0.5 d/kg. Measurements taken for voles have demonstrated a transfer factor for lead of 0.14 d/kg (Thomas 1997). The difference in the transfer factor for small mammal (0.5 d/kg) compared to that for beef (0.0004 d/kg) demonstrates that, although there is uncertainty in the scaling, there is a need to adjust the transfer factors, particularly when the body size are significantly different. Using the same extrapolation approach a transfer factor for radium uptake by a vole can be calculated to be 2.4 d/kg compared to a measured value of 2.5 d/kg (Thomas 1997). Information on the uptake of contaminants by small mammals can also be obtained from Sample *et al.* (1998), which provides concentration ratios for different types of small mammals (herbivore, insectivore, omnivore and general). Using the characteristics of a vole the transfer factor can be estimated from information provided by Sample *et al.* (1998) and compared to the predictions from the allometric scaling. Many of the contaminants (e.g., Fe, Se) are shown to be well-represented by the predicted transfer factor using allometric scaling.

Due to the lack of available transfer factors for different non-human biota for all elements, it is recommended that allometric scaling of the transfer factors based on beef is used. This approach provides a reasonable estimate of the transfer factor that can be used to calculate the concentration of non-human biota.

When the allometric equations for the transfer factor are combined with the allometric equation for food intake, it results in generally a constant trans-species ratio between whole-body and dietary activity concentrations (e.g., Beresford 2007). This approach was used in FASSET (2003). This approach would be appropriate however, as data in literature is generally reported as transfer factors, as opposed to concentration ratios, the use of the selected approach allowed a larger database to be selected.

Feed-to-beef transfer factors from literature were used as a starting point and model outputs for predicted concentrations in lemming under baseline conditions were compared with the measured site data for lemming (Table 3.2-13). A similar comparison was made for model outputs for predicted concentrations in caribou under baseline conditions and the measured caribou data from the site. Modifications were made to the feed-to-beef transfer factors from literature to achieve comparable concentrations in the lemming and caribou. The modifications for lemming were applied to the other small mammals, such as shrew and squirrel, while the modifications for caribou were applied to the large animals, such as bear, fox, muskox, wolf, and wolverine. The modified feed-to-beef transfer factors were scaled by body weight for each mammal VEC using equation (5.1-3).

$$TF_{VEC} = TF_{beef} \times \left(\frac{BW_{VEC}}{BW_{beef}} \right)^{-0.75} \qquad (5.1\text{-}3)$$

Table 5.1-10 presents the calibrated feed-to-beef transfer factors used in the assessment. Lognormal probabilistic distributions were derived using an assumed geometric standard deviation of 2.5. Attachment E provides a comparison of the model calculated lemming and caribou concentrations with measured lemming and caribou concentrations from the site.

Table 5.1-10 Summary of Feed-to-Mammal Transfer Factors

Constituent	Geometric Mean	GSD	Min	Max
LogNormal Distribution for Feed-to-Bear Transfer Factors (d/g ww)				
Uranium	4.0×10^{-6}	2.5	2.5×10^{-7}	6.2×10^{-5}
Thorium-230	1.0×10^{-6}	2.5	6.4×10^{-8}	1.6×10^{-5}
Lead-210	1.4×10^{-7}	2.5	9.1×10^{-9}	2.2×10^{-6}
Radium-226	1.1×10^{-6}	2.5	7.3×10^{-8}	1.8×10^{-5}
Polonium-210	4.1×10^{-6}	2.5	2.6×10^{-7}	6.5×10^{-5}
Arsenic	1.3×10^{-5}	2.5	8.1×10^{-7}	2.0×10^{-4}
Cadmium	2.4×10^{-6}	2.5	1.6×10^{-7}	3.8×10^{-5}
Cobalt	1.6×10^{-6}	2.5	1.0×10^{-7}	2.6×10^{-5}
Copper	4.5×10^{-5}	2.5	2.9×10^{-6}	7.0×10^{-4}
Lead	1.0×10^{-6}	2.5	6.6×10^{-8}	1.6×10^{-5}
Molybdenum	4.7×10^{-6}	2.5	3.0×10^{-7}	7.3×10^{-5}
Nickel	9.8×10^{-7}	2.5	6.3×10^{-8}	1.5×10^{-5}
Selenium	4.4×10^{-5}	2.5	2.8×10^{-6}	6.9×10^{-4}

Table 5.1-10 Summary of Feed-to-Mammal Transfer Factors

Constituent	Geometric Mean	GSD	Min	Max
Zinc	$4.7x10^{-5}$	2.5	$3.0x10^{-6}$	$7.4x10^{-4}$
LogNormal Distribution for Feed-to-Caribou Transfer Factors (d/g ww)				
Uranium	$7.2x10^{-6}$	2.5	$4.6x10^{-7}$	$1.1x10^{-4}$
Thorium-230	$1.8x10^{-6}$	2.5	$1.2x10^{-7}$	$2.9x10^{-5}$
Lead-210	$2.6x10^{-7}$	2.5	$1.7x10^{-8}$	$4.1x10^{-6}$
Radium-226	$2.1x10^{-6}$	2.5	$1.3x10^{-7}$	$3.2x10^{-5}$
Polonium-210	$7.5x10^{-6}$	2.5	$4.8x10^{-7}$	$1.2x10^{-4}$
Arsenic	$2.3x10^{-5}$	2.5	$1.5x10^{-6}$	$3.6x10^{-4}$
Cadmium	$4.4x10^{-6}$	2.5	$2.8x10^{-7}$	$6.9x10^{-5}$
Cobalt	$3.0x10^{-6}$	2.5	$1.9x10^{-7}$	$4.6x10^{-5}$
Copper	$8.2x10^{-5}$	2.5	$5.2x10^{-6}$	$1.3x10^{-3}$
Lead	$1.9x10^{-6}$	2.5	$1.2x10^{-7}$	$2.9x10^{-5}$
Molybdenum	$8.5x10^{-6}$	2.5	$5.4x10^{-7}$	$1.3x10^{-4}$
Nickel	$1.8x10^{-6}$	2.5	$1.1x10^{-7}$	$2.8x10^{-5}$
Selenium	$8.0x10^{-5}$	2.5	$5.1x10^{-6}$	$1.2x10^{-3}$
Zinc	$8.6x10^{-5}$	2.5	$5.5x10^{-6}$	$1.3x10^{-3}$
LogNormal Distribution for Feed-to-Fox Transfer Factors (d/g ww)				
Uranium	$1.0x10^{-4}$	2.5	$6.4x10^{-6}$	$1.6x10^{-3}$
Thorium-230	$2.6x10^{-5}$	2.5	$1.6x10^{-6}$	$4.0x10^{-4}$
Lead-210	$3.6x10^{-6}$	2.5	$2.3x10^{-7}$	$5.7x10^{-5}$
Radium-226	$2.9x10^{-5}$	2.5	$1.9x10^{-6}$	$4.5x10^{-4}$
Polonium-210	$1.1x10^{-4}$	2.5	$6.7x10^{-6}$	$1.6x10^{-3}$
Arsenic	$3.2x10^{-4}$	2.5	$2.1x10^{-5}$	$5.1x10^{-3}$
Cadmium	$6.2x10^{-5}$	2.5	$4.0x10^{-6}$	$9.7x10^{-4}$
Cobalt	$4.2x10^{-5}$	2.5	$2.7x10^{-6}$	$6.5x10^{-4}$
Copper	$1.1x10^{-3}$	2.5	$7.3x10^{-5}$	$1.8x10^{-2}$
Lead	$2.6x10^{-5}$	2.5	$1.7x10^{-6}$	$4.1x10^{-4}$
Molybdenum	$1.2x10^{-4}$	2.5	$7.6x10^{-6}$	$1.9x10^{-3}$
Nickel	$2.5x10^{-5}$	2.5	$1.6x10^{-6}$	$3.9x10^{-4}$
Selenium	$1.1x10^{-3}$	2.5	$7.2x10^{-5}$	$1.7x10^{-2}$
Zinc	$1.2x10^{-3}$	2.5	$7.7x10^{-5}$	$1.9x10^{-2}$
LogNormal Distribution for Feed-to-Lemming Transfer Factors (d/g ww)				
Uranium	$9.8x10^{-4}$	2.5	$6.3x10^{-5}$	$1.5x10^{-2}$
Thorium-230	$6.3x10^{-3}$	2.5	$4.0x10^{-4}$	$9.8x10^{-2}$
Lead-210	$8.5x10^{-3}$	2.5	$5.4x10^{-4}$	$1.3x10^{-1}$
Radium-226	$8.4x10^{-3}$	2.5	$5.4x10^{-4}$	$1.3x10^{-1}$
Polonium-210	$1.3x10^{-3}$	2.5	$8.5x10^{-5}$	$2.1x10^{-2}$

Table 5.1-10 Summary of Feed-to-Mammal Transfer Factors

Constituent	Geometric Mean	GSD	Min	Max
Arsenic	$6.0x10^{-3}$	2.5	$3.8x10^{-4}$	$9.3x10^{-2}$
Cadmium	$9.6x10^{-3}$	2.5	$6.1x10^{-4}$	$1.5x10^{-1}$
Cobalt	$9.1x10^{-3}$	2.5	$5.8x10^{-4}$	$1.4x10^{-1}$
Copper	$2.2x10^{-2}$	2.5	$1.4x10^{-3}$	$3.4x10^{-1}$
Lead	$3.7x10^{-3}$	2.5	$2.4x10^{-4}$	$5.8x10^{-2}$
Molybdenum	$5.4x10^{-3}$	2.5	$3.5x10^{-4}$	$8.4x10^{-2}$
Nickel	$3.7x10^{-3}$	2.5	$2.3x10^{-4}$	$5.7x10^{-2}$
Selenium	$6.3x10^{-3}$	2.5	$4.0x10^{-4}$	$9.8x10^{-2}$
Zinc	$2.9x10^{-2}$	2.5	$1.9x10^{-3}$	$4.5x10^{-1}$
LogNormal Distribution for Feed-to-Muskox Transfer Factors (d/g ww)				
Uranium	$4.0x10^{-6}$	2.5	$2.5x10^{-7}$	$6.2x10^{-5}$
Thorium-230	$1.0x10^{-6}$	2.5	$6.4x10^{-8}$	$1.6x10^{-5}$
Lead-210	$1.4x10^{-7}$	2.5	$9.1x10^{-9}$	$2.2x10^{-6}$
Radium-226	$1.1x10^{-6}$	2.5	$7.3x10^{-8}$	$1.8x10^{-5}$
Polonium-210	$4.1x10^{-6}$	2.5	$2.6x10^{-7}$	$6.5x10^{-5}$
Arsenic	$1.3x10^{-5}$	2.5	$8.1x10^{-7}$	$2.0x10^{-4}$
Cadmium	$2.4x10^{-6}$	2.5	$1.6x10^{-7}$	$3.8x10^{-5}$
Cobalt	$1.6x10^{-6}$	2.5	$1.0x10^{-7}$	$2.6x10^{-5}$
Copper	$4.5x10^{-5}$	2.5	$2.9x10^{-6}$	$7.0x10^{-4}$
Lead	$1.0x10^{-6}$	2.5	$6.6x10^{-8}$	$1.6x10^{-5}$
Molybdenum	$4.7x10^{-6}$	2.5	$3.0x10^{-7}$	$7.3x10^{-5}$
Nickel	$9.8x10^{-7}$	2.5	$6.3x10^{-8}$	$1.5x10^{-5}$
Selenium	$4.4x10^{-5}$	2.5	$2.8x10^{-6}$	$6.9x10^{-4}$
Zinc	$4.7x10^{-5}$	2.5	$3.0x10^{-6}$	$7.4x10^{-4}$
LogNormal Distribution for Feed-to-Shrew Transfer Factors (d/g ww)				
Uranium	$8.5x10^{-3}$	2.5	$5.4x10^{-4}$	$1.3x10^{-1}$
Thorium-230	$5.4x10^{-2}$	2.5	$3.5x10^{-3}$	$8.5x10^{-1}$
Lead-210	$7.3x10^{-2}$	2.5	$4.7x10^{-3}$	1.1
Radium-226	$7.3x10^{-2}$	2.5	$4.7x10^{-3}$	1.1
Polonium-210	$1.2x10^{-2}$	2.5	$7.4x10^{-4}$	$1.8x10^{-1}$
Arsenic	$5.2x10^{-2}$	2.5	$3.3x10^{-3}$	$8.1x10^{-1}$
Cadmium	$8.3x10^{-2}$	2.5	$5.3x10^{-3}$	1.3
Cobalt	$7.9x10^{-2}$	2.5	$5.0x10^{-3}$	1.2
Copper	$1.9x10^{-1}$	2.5	$1.2x10^{-2}$	2.9
Lead	$3.2x10^{-2}$	2.5	$2.1x10^{-3}$	$5.0x10^{-1}$
Molybdenum	$4.7x10^{-2}$	2.5	$3.0x10^{-3}$	$7.3x10^{-1}$
Nickel	$3.2x10^{-2}$	2.5	$2.0x10^{-3}$	$4.9x10^{-1}$

Table 5.1-10 Summary of Feed-to-Mammal Transfer Factors

Constituent	Geometric Mean	GSD	Min	Max
Selenium	$5.4x10^{-2}$	2.5	$3.5x10^{-3}$	$8.5x10^{-1}$
Zinc	$2.5x10^{-1}$	2.5	$1.6x10^{-2}$	3.9
LogNormal Distribution for Feed-to-Squirrel Transfer Factors (d/g ww)				
Uranium	$1.6x10^{-4}$	2.5	$1.0x10^{-5}$	$2.5x10^{-3}$
Thorium-230	$1.0x10^{-3}$	2.5	$6.5x10^{-5}$	$1.6x10^{-2}$
Lead-210	$1.4x10^{-3}$	2.5	$8.8x10^{-5}$	$2.1x10^{-2}$
Radium-226	$1.4x10^{-3}$	2.5	$8.8x10^{-5}$	$2.1x10^{-2}$
Polonium-210	$2.2x10^{-4}$	2.5	$1.4x10^{-5}$	$3.4x10^{-3}$
Arsenic	$9.7x10^{-4}$	2.5	$6.2x10^{-5}$	$1.5x10^{-2}$
Cadmium	$1.6x10^{-3}$	2.5	$1.0x10^{-4}$	$2.4x10^{-2}$
Cobalt	$1.5x10^{-3}$	2.5	$9.5x10^{-5}$	$2.3x10^{-2}$
Copper	$3.5x10^{-3}$	2.5	$2.2x10^{-4}$	$5.5x10^{-2}$
Lead	$6.0x10^{-4}$	2.5	$3.9x10^{-5}$	$9.4x10^{-3}$
Molybdenum	$8.8x10^{-4}$	2.5	$5.6x10^{-5}$	$1.4x10^{-2}$
Nickel	$5.9x10^{-4}$	2.5	$3.8x10^{-5}$	$9.3x10^{-3}$
Selenium	$1.0x10^{-3}$	2.5	$6.5x10^{-5}$	$1.6x10^{-2}$
Zinc	$4.7x10^{-3}$	2.5	$3.0x10^{-4}$	$7.4x10^{-2}$
LogNormal Distribution for Feed-to-Wolf Transfer Factors (d/g ww)				
Uranium	$1.7x10^{-5}$	2.5	$1.1x10^{-6}$	$2.7x10^{-4}$
Thorium-230	$4.3x10^{-6}$	2.5	$2.8x10^{-7}$	$6.8x10^{-5}$
Lead-210	$6.1x10^{-7}$	2.5	$3.9x10^{-8}$	$9.6x10^{-6}$
Radium-226	$4.9x10^{-6}$	2.5	$3.1x10^{-7}$	$7.7x10^{-5}$
Polonium-210	$1.8x10^{-5}$	2.5	$1.1x10^{-6}$	$2.8x10^{-4}$
Arsenic	$5.5x10^{-5}$	2.5	$3.5x10^{-6}$	$8.5x10^{-4}$
Cadmium	$1.0x10^{-5}$	2.5	$6.7x10^{-7}$	$1.6x10^{-4}$
Cobalt	$7.0x10^{-6}$	2.5	$4.5x10^{-7}$	$1.1x10^{-4}$
Copper	$1.9x10^{-4}$	2.5	$1.2x10^{-5}$	$3.0x10^{-3}$
Lead	$4.4x10^{-6}$	2.5	$2.8x10^{-7}$	$6.9x10^{-5}$
Molybdenum	$2.0x10^{-5}$	2.5	$1.3x10^{-6}$	$3.1x10^{-4}$
Nickel	$4.2x10^{-6}$	2.5	$2.7x10^{-7}$	$6.6x10^{-5}$
Selenium	$1.9x10^{-4}$	2.5	$1.2x10^{-5}$	$2.9x10^{-3}$
Zinc	$2.0x10^{-4}$	2.5	$1.3x10^{-5}$	$3.2x10^{-3}$
LogNormal Distribution for Feed-to-Wolverine Transfer Factors (d/g ww)				
Uranium	$4.2x10^{-5}$	2.5	$2.7x10^{-6}$	$6.5x10^{-4}$
Thorium-230	$1.1x10^{-5}$	2.5	$6.8x10^{-7}$	$1.7x10^{-4}$
Lead-210	$1.5x10^{-6}$	2.5	$9.6x10^{-8}$	$2.3x10^{-5}$
Radium-226	$1.2x10^{-5}$	2.5	$7.7x10^{-7}$	$1.9x10^{-4}$

Table 5.1-10 Summary of Feed-to-Mammal Transfer Factors

Constituent	Geometric Mean	GSD	Min	Max
Polonium-210	$4.4x10^{-5}$	2.5	$2.8x10^{-6}$	$6.8x10^{-4}$
Arsenic	$1.3x10^{-4}$	2.5	$8.6x10^{-6}$	$2.1x10^{-3}$
Cadmium	$2.6x10^{-5}$	2.5	$1.6x10^{-6}$	$4.0x10^{-4}$
Cobalt	$1.7x10^{-5}$	2.5	$1.1x10^{-6}$	$2.7x10^{-4}$
Copper	$4.7x10^{-4}$	2.5	$3.0x10^{-5}$	$7.4x10^{-3}$
Lead	$1.1x10^{-5}$	2.5	$6.9x10^{-7}$	$1.7x10^{-4}$
Molybdenum	$4.9x10^{-5}$	2.5	$3.1x10^{-6}$	$7.7x10^{-4}$
Nickel	$1.0x10^{-5}$	2.5	$6.6x10^{-7}$	$1.6x10^{-4}$
Selenium	$4.6x10^{-4}$	2.5	$3.0x10^{-5}$	$7.2x10^{-3}$
Zinc	$5.0x10^{-4}$	2.5	$3.2x10^{-5}$	$7.8x10^{-3}$

5.1.4 Dose Coefficients

5.1.4.1 Human

The assessment used the dose coefficients (DCs) recommended by the International Commission on Radiological Protection (ICRP). Ingestion DCs depend on the chemical form of the radionuclide and the consequent gut-to-blood transfer factor (f1). Table 5.1-10 reflects the ICRP Publication 72 (1996) recommended f1 values and DCs for members of the public. Inhalation DCs depend on the chemical form of the radionuclide and the consequent rate of clearance from the lungs to body fluids - slow (S), moderate (M) or fast (F). The ICRP recommends type M for most unspecified conditions with the exception of thorium-230 for which type S is recommended. To be conservative, the generally larger DCs (i.e., less soluble S type DCs) were used for all radionuclides in this assessment.

Dose coefficients for exposure through inhalation and ingestion are shown in Table 5.1-11.

Table 5.1-11 Dose Coefficients Used for Human Receptors

Constituent	Toddler [a]		Child [b]		Adult	
	Inhalation	Ingestion	Inhalation	Ingestion	Inhalation	Ingestion
Uranium-nat	52.5	0.25	36	0.185	17.8	0.0995
Thorium-230	35	0.41	24	0.31	14	0.21
Lead-210+	18.3	3.6	11	2.2	5.6	0.69
Radium-226+	29	0.96	19	0.62	9.5	0.28
Polonium-210	14	8.8	8.6	4.4	4.3	1.2

Note:
Units: µSv/Bq
References: ICRP 72 (1996)
a Taken to be equivalent to a 1-year old
b Taken to be equivalent to a 5-year old

5.1.4.2 Ecological

Table 5.1-12 shows the selected DCs for the estimation of dose to ecological receptors. The DCs were obtained from Amiro (1997); DCs for internal and external exposure are provided.

Table 5.1-12 Dose Coefficients Used for Ecological Receptors

COPC[b]	DC (in mGy/d per Bq/g)[a]	
	Internal	External[c]
Uranium-238+	1.42×10^{-1}	8.57×10^{-3}
Thorium-230	6.58×10^{-2}	2.93×10^{-5}
Lead-210+	6.03×10^{-3}	2.50×10^{-3}
Radium-226+	1.59×10^{-1}	3.68×10^{-2}
Polonium-210	7.40×10^{-2}	1.41×10^{-7}

Reference:

a Dose (mGy/d) per Bq/g tissue concentration for internal dose and per Bq/g of environmental concentration for external dose

b The radionuclides included in each dose coefficient (DC) for decay series were derived from the DCs in Amiro (1997) as follows

Uranium-238+ = U-238 + Th-234 + U-234 + 0.046 U-235 (internal or external)

Radium-226+ = Ra-226 + 0.3* Rn-222 (internal) where 0.3 = fraction of Rn-222 retained in organism (Blaylock et al. 1993)

= Ra-226 + 1.0* Rn-222 (external)

Lead-210 = Pb-210 + Bi-210 (internal or external)

c For external dose, the external DCs from Amiro (1997) for soil were used.

d In Amiro (1997), Th-234 = Pa-234m + 0.0012*Pa-234; and Rn-222 = Po-218 + Pb-214 + Bi-214 + Po-214

5.2 Predicted Intakes

The predicted intakes by terrestrial biota and human receptors are provided in Attachment F. These intakes are considered with the toxicity data to evaluate the potential for negative impacts from the Project. The predicted intakes show that, in general, for ecological receptors, there is not much change in intakes over the course of the assessment period, which indicates that there is not much noticeable difference from the Project activities. However, receptors at the Kiggavik Camp and discharge locations to Judge Sissons Lake do occasionally show an increase in intakes during the operating phase of the Project.

For human receptors, the predicted intakes show that the Judge Sissons Lake hunter and family are the most exposed to effects from the Project, with the Baker Lake residents and non-NEW Camp worker less exposed.

6 Toxicitiy Assessment

This section outlines the assessment endpoints for the ecological and human health risk assessment. The assessment endpoints, or toxicity reference values (TRVs), selected for aquatic species and wildlife are considered threshold concentrations or doses/intakes of the non-radioactive and radioactive constituents that can cause harm in ecological species (i.e., VECs). The TRVs for humans are intended to protect the most sensitive individuals (i.e., the elderly, pregnant women and children). For COPC that have non-carcinogenic effects, the TRVs are based on threshold effects concentrations. For COPC that have carcinogenic effects (i.e., those COPC that can cause cancer), the TRVs are based on non-threshold effects.

For the radiological COPC, the effects assessment is based on total exposures (dose rates) for VECs. Humans are evaluated on an incremental exposure basis.

6.1 Criteria for Water Quality

The potential impacts of the project on water quality in the receiving environment were evaluated by comparing predicted COPC concentrations in water to available water quality guidelines (WQGs). Selected water quality guidelines are the lowest of either Canadian Water Quality Guidelines for the Protection of Aquatic Life (CCME 2012) or Health Canada Guidelines for Canadian Drinking Water Quality (HC 2010a). If a value was not available from these sources then other jurisdictions were also considered, for example the water quality guidelines derived by the British Columbia Ministry of Environment or Saskatchwan Environment. Available WQGs for the protection of aquatic life for the COPCs are summarized in Table 6.1-1.

Table 6.1-1 Water Quality Objectives and Guidelines

COPC	WQG	Units	Note
Arsenic	5	µg/L	CWQG
Cadmium	0.04-0.37	µg/L	CWQG: 0.04 ug/L at hardness between 0 and 17 mg/L; at hardness between 17 mg/L and 280 mg/L = $10^{\{0.83(\log[\text{hardness}]) - 2.46\}}$; 0.37 ug/L at hardness greater than 280 mg/L
Cobalt	4	µg/L	BCWQG
Copper	2-4	µg/L	CWQG: 2 ug/L at hardness between 0 and 82 mg/L; at hardness between 82 mg/L and 180 mg/L = $0.2 \times e^{\{0.8545(\ln[\text{hardness}]) - 1.465\}}$; 4 ug/L at hardness greater than 180 mg/L
Lead	1-7	µg/L	CWQG: 1 ug/L at hardness between 0 and 60 mg/L; at hardness between 60 mg/L and 180 mg/L = $e^{\{1.273(\ln[\text{hardness}]) - 4.705\}}$; 7 ug/L at hardness greater than 180 mg/L
Molybdenum	73	µg/L	Interim CWQG

Table 6.1-1 Water Quality Objectives and Guidelines

COPC	WQG	Units	Note
Nickel	25-150	µg/L	CWQG: 25 ug/L at hardness between 0 and 60 mg/L; at hardness between 60 mg/L and 180 mg/L = $e^{\{0.76(\ln[\text{hardness}]) - 1.06\}}$; 150 ug/L at hardness greater than 180 mg/L
Selenium	1	µg/L	CWQG
Uranium	15	µg/L	CWQG
Zinc	30	µg/L	CWQG
Thorium-230	0.6	Bq/L	CDWQG (Appendix A, Health Canada 2010a)
Lead-210	0.2	Bq/L	CDWQG
Radium-226	0.11	Bq/L	Historical SSWQO, retained in absence of other guidelines
Polonium-210	0.1	Bq/L	CDWQG (Appendix A, Health Canada 2010a)
Un-ionized Ammonia	0.019	mg/L	CWQG for total ammonia 1.54 mg/L-NH_3 converted to mg/L-N using historic measurements of temperature and pH (lower pH values and temperatures would result in a high allowable ammonia concentration)
Chloride	120	mg/L	CWQG
Sulphate	128 218 309	mg/L	BCWQG for very soft water (0-30 mg/L) BCWQG for soft water (31-75 mg/L) BCWQG for moderate water (76-180 mg/L)
TDS	500	mg/L	CDWQG
Water Hardness	-	-	No guideline available

Note:
SSWQO – Saskatchewan Surface Water Quality Objective for the Protection of Aquatic Life (Saskatchewan Environment 2006)
BCWQG – British Columbia Water Quality Guideline (BC MOE 2014)
CWQG – Canadian Water Quality Guideline (CCME 2013)
CDWQG – Canadian Drinking Water Quality Guideline (Health Canada 2010a)

6.2 Assessment Criteria for Sediment Quality

The potential ecological effects of sediment contamination at the Kiggavik Project site were addressed in part through the examination of potential effects on benthic invertebrates. In contrast to the approach outlined above to assess the ecological risks to aquatic species from exposure to COPC in the water column, the sediment toxicity evaluation involved comparison of measured and/or predicted levels of COPC in sediments to sediment quality guidelines.

Table 6.2-1 outlines theoretical toxicity benchmarks from Thompson *et al*. (2005) that are specific to uranium-bearing regions of Canada (e.g., northern Saskatchewan and northern Ontario) and are considered Canadian Nuclear Safety Commission (CNSC) working reference values. They have proposed benchmarks for the radionuclides of interest in the current assessment and for most of the metals; however, no sediment toxicity benchmarks exist for cadmium, cobalt, zinc, ammonia,

sulphate, or thorium-230. Table 6.2-1 also provides the available sediment quality guidelines from CCME (2013) as well as the results of toxicity testing on the growth and reproduction of *Hyalella azteca* and *Chironomus dilutus* (Liber et al. 2011). The potential adverse effects of thorium-230 in sediment on benthic organisms were considered through the radiological assessment, since there are no available sediment quality guidelines.

Table 6.2-1 Sediment Quality Toxicity Benchmarks

COPC	Sediment Quality Guidelines					
	Thompson et al. (2005)		CCME (2013)		Liber et al. 2011	
	LEL	SEL	ISQG	PEL		
Metal (µg/g)						
Arsenic	9.8	346.4	5.9	17	174	IC_{25} growth
Cadmium	-	-	0.6	3.5		
Cobalt	-	-	-	-	-	-
Copper	22.2	268.8	35.7	197	-	-
Lead	36.7	412.4	35	91.3	-	-
Molybdenum	13.8	1238.5	-	-	3589	NOEC
Nickel	23.4	484	-	-	189	IC_{25} growth
Selenium	1.9	16.1	-	-	-	-
Uranium	104.4	5874.1	-	-	964	IC_{25} growth
Zinc	-	-	123	315	-	-
Radionuclide (Bq/g)						
Thorium-230	-	-	-	-	-	-
Radium-226	0.6	14.4	-	-	-	-
Lead-210	0.9	20.8	-	-	-	-
Polonium-210	0.8	12.1	-	-	-	-

Note: - - not available
LEL - lowest effect level
SEL - severe effect level
ISQG - Interim Sediment Quality Guideline
PEL - probable effect level
IC_{25} - - inhibition concentration (e.g., 25% decrease in growth)
NOEC – No Observed Effect Concentration

As seen from Table 6.2-1, Thompson *et al.* (2005) have proposed both Lowest Effect Level (LEL) and Severe Effect Level (SEL) for benchmarks. The LEL represents the COPC concentration below which harmful effects on benthic invertebrates are not expected. In a recent evaluation of data from northern Saskatchewan uranium operations and the sediment objectives, it was shown that although LELs generally align correctly with effects observations at 95% of sites that had effects, it was also noted that exceedances of LEL values were also observed at 60% of the no-effect sites (Burnett-Seidel and Liber 2012). The SEL represents the COPC concentration above which harmful effects

on benthic invertebrates are expected. Two statistical methods were used by Thompson *et al*. (2005) to define the percentiles corresponding to LEL and SEL concentrations. A "weighted method" produced somewhat higher values than a "closest observation method". When the predictive ability of the sediment quality guidelines was assessed, all of the LELs derived using the weighted method, with the exception of the chromium LEL, were found to be highly reliable (> 85% accuracy) in predicting sites unimpacted by uranium mining/milling. Caution is to be employed when using the SEL values as they may not be a reliable predictor of potential effects. A weight-of-evidence approach where the use of these sediment quality benchmarks and actual data collected in the field is provided in this assessment.

6.3 Assessment Criteria for Ecological Toxicity

By convention, the assessment of effects of non-radiological COPC considers total exposure concentrations (levels) for aquatic VECs and total intakes for terrestrial VECs. Within an ecological risk assessment framework, VEC assessment endpoints represent biological populations and communities to be protected. Hence, TRVs are used in risk assessments to judge whether the predicted (estimated) exposures (or doses or intakes) may potentially have an adverse effect on ecological species at the population level as opposed to individual members of the species. An exception to this general rule applies to rare or endangered species where an adverse effect at the individual level represents a concern.

The TRVs for aquatic species are generally based on acute and/or chronic toxicity tests carried out under standardized conditions in the laboratory using sensitive test species (e.g., rainbow trout). There is a lack of data on the individual terrestrial mammals; in the absence of species-specific toxicity data, data for laboratory animals (often mice and rats) are selected. For avian receptors, the test species are generally ducks or chickens for which large datasets are available. The background information for the toxicity reference values selected for this assessment are provided in Attachment G. The TRVs selected for use for the Kiggavik Project are consistent with those used in a recently completed assessment for the water management project at the Cigar Lake uranium mine in northern Saskatchewan (Cameco 2010).

6.3.1 Non-radioactive Toxicity Reference Values for Aquatic Species

The U.S. EPA ECOTOXicology database (ECOTOX) (U.S. EPA 2000) was the primary source used for the derivation of the TRVs for aquatic biota. This database reports toxicity data for a wide range of aquatic species as well as laboratory and field studies. Other sources included the Environment Canada Priority Substance Lists, the datasets underlying the Canadian Council of Ministers of the Environment (CCME) Water Quality Guidelines, U.S. Environmental Protection Agency (U.S.EPA) Water Quality Guidelines and recent compilations of aquatic toxicity information (e.g., USGS 2010). For most non-radionuclides, ECOTOX includes toxicity data in literature from 1972 to the present. All data are quality assured according to the EPA's criteria, and the system is updated quarterly.

For the purpose of this assessment, the following principles were applied in the data selection from ECOTOX.

- Endpoints involving growth, reproduction and survival were considered to be relevant to the persistence of aquatic populations.

- Only freshwater toxicity studies were considered.

- Records without test duration, endpoint and exposure concentrations were eliminated.

- Chronic toxicity data were preferred in the selection. When chronic data were not sufficient (less than of 2 data points), acute data were considered and converted to chronic values. The adjustment from acute LC50 to chronic EC20 (a 1/10 factor) was used based on the work by Environment Canada and Health Canada (EC/HC 2003) from a review of aquatic toxicity literature for uranium.

- Chronic EC20 concentrations were preferred. If not reported, other endpoints were considered and adjusted to an estimated EC20 value. For adjustment from chronic LC50 to chronic EC20, a ¼ factor was used based on an assumed linear chronic dose-response with zero response at EC0 and 50% response at the EC_{50} concentration.

The TRVs derived for aquatic biota included five categories: forage fish, predator fish, aquatic plants, zooplankton and benthic invertebrates. If more than 20 chronic EC_{20} were available in each category, a 5th percentile of the EC_{20} distribution was used as a recommended TRV; if there were less than 20 chronic EC_{20} values, the lowest EC_{20} was used as a recommended TRV for the category. The recommended TRVs for each category in this review are summarized in Table 6.3-1. As seen from Table 6.3-1, the TRVs for aquatic biota are based on water concentrations; the exception to this is selenium, where the TRVs for fish are based on tissue concentrations since selenium is known to bioconcentrate in the aquatic environment. In a few cases, the TRVs were set equal to the Canadian Water Quality Guideline (CWQG) to avoid unrealistic values. Additional details on the basis of the TRVs are provided in Attachment G. It should be noted that TDS is an integrative constituent which is used as a measure of common ions such as sulphate and chloride in water. TDS is not a robust predictor of toxicity for a range of water qualities (Mount et al. 1997) and thus its toxicity is implicitly evaluated for chloride and sulphate in this assessment. Hardness was included in the modelling; it is not considered a COPC but affects the toxicity of other parameters. For example, the toxicity of several metals is dependent on hardness with decreasing toxicity at higher hardness levels.

Table 6.3-1 Selected Toxicity Reference Values for Aquatic Biota

COPC	Units	Forage Fish	Predator Fish	Aquatic Plants	Phyto-plankton	Zoo-plankton	Benthic Invertebrates
Arsenic	µg/L	123	630	252	5 [(c)]	340	122
Cadmium	µg/L	7.3	0.6	16	3	0.07	0.5
Cobalt	µg/L	203	118	54.3	212	4.8	4
Copper [(a)]	µg/L	6	4	38	9.2	4	4
Lead [(a)]	µg/L	132	14.2	439	114	40	1 [(d)]
Molybdenum	µg/L	5,000	183	15,000	15,000	233	250
Nickel [(a)]	µg/L	535	62	84	93.2	38	53.5
Selenium	µg/L	[(b)]	[(b)]	680	31.7	10	10
Uranium	µg/L	550	1,500	5,500	400	60	27
Zinc [(a)]	µg/L	35	30 [(e)]	116	30 [(e)]	30 [(e)]	30 [(e)]
Ammonia (un-ionized)	µg/L	173	90	-	800	160	120
Chloride	mg/L	220	360	1260	590	290	143
Sulphate	mg/L	501	933	828	1112	246	1056

Note:
a = Toxicity reference values (TRVs) based on low hardness water.
b = TRV for selenium based on 10 µg/g (dw) whole body fish concentration.
c = TRV set equal to CWQG for arsenic.
d = TRV set equal to CWQG for lead in low hardness water.
e = TRV set equal to CWQG for zinc.
- = no data available.
COPC = constituents of potential concern; µg/L = micrograms per litre

It is acknowledged that there is uncertainty associated with these values. The use of a range of biota in the assessment adds to the robustness of the overall assessment. It should be noted that within the risk assessment different endpoints are considered compared to those for generic guidelines. As discussed above and consistent with N288.6 (CSA 2012), benchmarks were selected that represent approximately an 20% effects level (EC_{20}). For example, in the development of the generic water quality guideline for uranium, the lowest value is for *Hyalella azteca*. According to CCME (2011), Liber et al. (2007) exposed *H. azteca* to uranium for 28 days in water and an EC_{50} of 67 µg/L and an EC_{25} of 27 µg/L were determined. The lower value was adopted for use in this assessment. CCME (2011) derived an EC_{10} for use in the development of a generic water quality guideline.

6.3.2 Non-radioactive Toxicity Reference Values for Terrestrial Species

6.3.2.1 Phytotoxicity

The literature has a number of studies relating to adverse effects in vegetation related to exposure to sulphur dioxide and nitrogen oxides.

Sulphur dioxide is known to have adverse effects on plants. Sulphur dioxide penetrates the stomata of plants. External factors such as humidity, temperature, and light intensity influence the rate of uptake of SO_2 and thus the degree of injury.

High concentrations of sulphur dioxide over a short duration of exposure are known to cause acute injury in the form of foliar necrosis. However, longer term effects of SO_2 exposure are more important since they occur at much lower concentrations and are cumulative in nature resulting in adverse effects such as reduced growth and yield and increased aging which may not be visible or manifested as chlorosis. Dose-response relationships have been developed for various agricultural crops and some temperate forest tree species but data on native herbaceous vegetation are scarce (WHO 2000). The current WHO phytotoxic guidelines for SO_2 are 20 µg/m^3 as an annual average for forests and natural vegetation (WHO 2000). A lower value of 10 µg/m^3 is also provided for lichen. A value of 100 µg/m^3 as a 24 hour average, although not officially adopted by the WHO was also used in the assessment (WHO 2000).

Nitrogen containing air pollutants can affect vegetation indirectly, via chemical reactions in the atmosphere or directly by deposition on vegetation soil or water. NO and NO_2 are precursors to trophospheric ozone which is also known to be phytotoxic. Uptake of NOx (NO and NO_2) in the leaves from wet deposition is generally via the cuticle. As with SO_2, environmental conditions such as humidity and temperature affect the phytotoxicity of NOx. The WHO (2000) suggests that threshold values are the best way to evaluate the phytotoxic effects of NOx. They indicate that a critical effects level (CLE) is a threshold that can be used to examine the direct effects of NOx since the CLE is the concentration in the atmosphere above which direct adverse effects on receptors such as plants and ecosystems may occur according to present information.

The WHO provides a CLE for NOx of 30 µg/m^3 on an annual basis. This level is used in the assessment. It was supported by the recent evaluation of nitrogen oxides by the US EPA that concluded that the existing secondary for oxides of nitrogen of 0.053 ppm (equivalent to 100 µg/m^3) are adequate to protect plants from the direct effects of oxides of nitrogen in the ambient air.

For metals, the plant concentration can be compared to levels known to illicit phytotoxicity. A summary of some phytotoxcity benchmarks is provided in Table 6.3-2.

Table 6.3-2 Summary of Phytotoxicity Levels

COPC	Units	Phytotoxic Leaf Conc.[a]	Phytotoxic foliage Conc.[b]	Upper Critical Level in leaves and shoots [c]
Arsenic	mg/kg dw	5-20	3-10	11-26
Cadmium	mg/kg dw	5-30	5-700	14-16
Cobalt	mg/kg dw	15-50	25-100	3-9
Copper	mg/kg dw	-	25-40	-
Lead	mg/kg dw	30-300	-	20-35
Molybdenum	mg/kg dw	10-50	100	130-140
Nickel	mg/kg dw	-	50-100	-
Selenium	mg/kg dw	5-30	100	7-90
Uranium	mg/kg dw	-	-	-
Zinc	mg/kg dw	100-400	500-1500	160-320

Note:

a = Leaf tissue concentration in plants that are neither sensitive or tolerant McBride, M.B. 1994 Environmental Chemistry of Soils. Oxford University Press Inc. New York, NY..

b =Phytotoxic conc. In plant foilage. Langmuir, D., P. Chrostowski, B. Vigneault and R. Chaney 2004. Issue Paper on the Environmental Chemistry of Metals. Submitted to U.S. Environmental Protection Agency, Risk Assessment Forum, Washington, DC. ERG, Lexington, MA

c = Upper Critical Level in leaves and shoots of spring barley associated with reduced yield. Davis, R.D., P.H.T. Beckett and E. Wollan 1978. Critical Levels of Twenty Potentially Toxic Elements in Young Spring Barley. Plant Soil 49: 395-408.

- = no data available.

COPC = constituents of potential concern; dw = dry weight

6.3.2.2 Wildlife

For the terrestrial species, the U.S. EPA risk-based ecological soil screening levels (Eco-SSLs) were used as the primary data source for the derivation of terrestrial TRVs. The Eco-SSLs were developed to limit the need by the EPA and other risk assessors to perform repetitious toxicity data literature searches and evaluations for the same COPC at every site. For this assessment, benchmark TRVs based on growth and reproduction were selected. Endpoints involving growth and reproduction are considered to be the most relevant endpoints related to maintenance and persistence of wildlife populations. Both no observable adverse effect levels (NOAELs) and lowest observable adverse effect levels (LOAELs) were selected for the assessment. NOAELs were only selected from the data if they had a corresponding LOAEL. All LOAELs with growth and reproductive endpoints were selected.

In the derivation of the TRVs, a number of guiding principles were followed as outlined below (with the exception of uranium).

- Studies were selected from the database where both NOAELs and LOAELs were reported as well as studies with only LOAELs. Studies that only reported NOAELs were not considered in the derivation of TRVs.

- A number of test species were selected to serve as surrogates for various wildlife species that are similar in size and diet. TRVs were calculated as the geometric mean of the selected NOAELs and LOAELs for several species for both avian and mammalian wildlife.

- The NOAEL TRV was derived from the geometric mean of the NOAEL data and the LOAEL TRV was derived from the geometric mean of the LOAEL data pooling growth and reproduction endpoints.

- NOAEL TRVs were only provided if there was an associated LOAEL TRV for the species; if only LOAELs were available then only a LOAEL TRV was provided.

For COPC without Eco-SSL data (in this case uranium), literature studies were reviewed and chronic LOAEL and NOAEL values were selected. Only growth and reproduction endpoints were considered, as described above for the Eco-SSL data. In general, TRVs from Sample et al. (1996) were used.

The Eco-SSL data base provides TRVs for a number of different species that could be used as a surrogate TRV for other species with similar diets. For example, for molybdenum the TRV for a calf was used as the surrogate for moose and caribou. If none of the test species were similar to the ecological receptors selected in this assessment, then the lowest NOAEL and LOAEL TRVs were selected as the conservative default TRV for the ecological receptor. For example, for zinc, no TRVs were available for species that had similar diets to the bear, fox, wolf, and wolverine; therefore, the lowest LOAEL that was developed for sheep was selected as the TRV for these species. Table 6.3-3 and Table 6.3-4 provide the TRVs selected for avian and mammalian receptors, respectively.

Table 6.3-3 Selected Toxicity Reference Values for Avian Species

COPC	Units	LOAEL		NOAEL	
		Waterfowl [(a)]	Terrestrial Birds [(b)]	Waterfowl [(a)]	Terrestrial Birds [(b)]
Arsenic	mg/(kg-d)	5.1	3.6	-	-
Cadmium	mg/(kg-d)	25.6	4.4	3.1	0.97
Cobalt	mg/(kg-d)	148	14.1	14.8	4.1
Copper	mg/(kg-d)	75.2	27	24.1	15.3
Lead	mg/(kg-d)	11.8*	11.8	-	-
Molybdenum	mg/(kg-d)	20.8*	20.8	-	-
Nickel	mg/(kg-d)	10.7	10.7	-	-
Selenium	mg/(kg-d)	1.37	0.59	1.18	-
Uranium	mg/(kg-d)	-	-	16	16
Zinc	mg/(kg-d)	62.7	62.7	-	-

Note:
a = Waterfowl (long-tailed duck, merganser, and pintail) toxicity reference values based on duck data when available.
b = Terrestrial birds include Falcon, Longspur, Ptarmigan, Sandpiper Eagle; toxicity reference values based lowest value available for all bird species.
COPC = constituents of potential concern; LOAEL = lowest observable adverse effect level; NOAEL = no observable adverse effect level
-: Not available; *: The lowest value (for all species) was used
mg/(kg-d) = milligrams per kilogram per day

Table 6.3-4 Selected Toxicity Reference Values for Mammalian Species

LOAEL / COPC	Unit	Bear	Caribou	Fox	Lemming	Muskox
Arsenic	mg/(kg-d)	3.1	14.4	3.1	14.2	14.4
Cadmium	mg/(kg-d)	100	5.7	100	6.8	5.7
Cobalt	mg/(kg-d)	13.4*	20.2	13.4*	13.4	20.2
Copper	mg/(kg-d)	11.5	1.5	11.5	194	1.5
Lead	mg/(kg-d)	50	15	50	188	15
Molybdenum	mg/(kg-d)	1.9*	4.1	1.9*	1.9	4.1
Nickel	mg/(kg-d)	112	9.8	112	11.6	9.8
Selenium	mg/(kg-d)	0.21	0.33	0.21	0.63	0.33
Uranium[a]	mg/(kg-d)	5.6*	5.6*	5.6*	5.6	5.6*
Zinc	mg/(kg-d)	34.9*	75.9	34.9*	249	75.9
COPC	**Unit**	**Shrew**	**Squirrel**	**Wolf**	**Wolverine**	
Arsenic	mg/(kg-d)	14.2	14.2	3.1	3.1	

Table 6.3-4 Selected Toxicity Reference Values for Mammalian Species

LOAEL COPC	Unit	Bear	Caribou	Fox	Lemming	Muskox
Cadmium	mg/(kg-d)	6.8	6.8	100	100	
Cobalt	mg/(kg-d)	13.4	13.4	13.4*	13.4*	
Copper	mg/(kg-d)	194	194	11.5	11.5	
Lead	mg/(kg-d)	188	188	50	50	
Molybdenum	mg/(kg-d)	1.9	1.9	1.9*	1.9*	
Nickel	mg/(kg-d)	11.6	11.6	112	112	
Selenium	mg/(kg-d)	0.63	0.63	0.21	0.21	
Uranium[a]	mg/(kg-d)	5.6	5.6	5.6*	5.6*	
Zinc	mg/(kg-d)	249	249	34.9*	34.9*	
NOAEL COPC	**Unit**	**Bear**	**Caribou**	**Fox**	**Lemming**	**Muskox**
Arsenic	mg/(kg-d)	1.5	-	1.5	8.1	-
Cadmium	mg/(kg-d)	8.3	-	8.3	1.4	-
Cobalt	mg/(kg-d)	5.2*	-	5.2*	5.2	-
Copper	mg/(kg-d)	5.9	-	5.9	100	-
Lead	mg/(kg-d)	-	-	-	25.5	-
Molybdenum	mg/(kg-d)	-	-	-	-	-
Nickel	mg/(kg-d)	45	3.4	45	6.2	3.4
Selenium	mg/(kg-d)	-	0.17	-	0.33	0.17
Uranium[a]	mg/(kg-d)	2.8*	2.8*	2.8*	2.8	2.8*
Zinc	mg/(kg-d)	-	37.9	-	215	37.9
COPC	**Unit**	**Shrew**	**Squirrel**	**Wolf**	**Wolverine**	
Arsenic	mg/(kg-d)	8.1	8.1	1.5	1.5	
Cadmium	mg/(kg-d)	1.4	1.4	8.3	8.3	
Cobalt	mg/(kg-d)	5.2	5.2	5.2*	5.2*	
Copper	mg/(kg-d)	100	100	5.9	5.9	
Lead	mg/(kg-d)	25.5	25.5	-	-	
Molybdenum	mg/(kg-d)	-	-	-	-	
Nickel	mg/(kg-d)	6.2	6.2	45	45	
Selenium	mg/(kg-d)	0.33	0.33	-	-	

NOAEL COPC	Unit	Bear	Caribou	Fox	Lemming
Uranium[a]	mg/(kg-d)	2.8	2.8	2.8*	2.8*

NOAEL COPC	Unit	Bear	Caribou	Fox	Lemming
Zinc	mg/(kg-d)	215	215	-	-

Note:

* = No surrogate species data available for these species therefore lowest TRV for other species used.

a = The Sample et al. data for mammalian species were based on a study by Paternain et al. (1989) related to reproduction in mice. Sample derived a NOAEL of 3.07 mg/kg/day and a LOAEL of 6.13 mg/kg/day from the study. However, the TRV quoted by Sample et al. (1996) contains a small unit conversion error. Instead of 6.13 mg /kg/d for the LOAEL as reported, the value should be 5.6 mg /kg/d. The lowest dose in the Paternain et al. study was 2.8 mg/kg/d which is considered to be the NOAEL. Total implantations differed between treatment and control at this dose only, but there was no dose response in this endpoint, and all parameters of reproductive success were unimpaired at this dose level. For this reason the lowest dose (2.8 mg/kg/d) is considered to be a NOAEL.

mg/(kg-d) = milligrams per kilogram per day

6.4 Assessment Criteria for Human Toxicity

To assess effects on human health, it is necessary to identify assessment endpoints by using toxicity reference values (TRVs) against which the effects will be measured. The background information used in the selection of the TRVs applied in this assessment is provided in Attachment G. Selected TRVs for assessment of oral exposure of human receptors to nonradiological and radiological COPC are discussed in the following subsections. The inhalation pathway is considered in the Project effects; however, this pathway is only considered in the total exposure evaluation, since the impacts from this pathway alone are very small.

6.4.1 Non-Radioactive Toxicity Reference Values for Human Health

The TRVs for humans are intended to protect the most sensitive individuals (i.e., the elderly, pregnant women and children). For COPC that have non-carcinogenic effects, the TRVs are based on threshold effects concentrations. For COPC that have carcinogenic effects (i.e., those COPC that can cause cancer), the TRVs are based on non-threshold effects.

The COPC selected for this Environmental Impact Statement (EIS) are, for the most part, considered to be non-classifiable with respect to human carcinogenicity (with respect to the oral exposure route), meaning that there are insufficient human or animal data to indicate that they are carcinogens. The exception to this is arsenic, which is considered to have both carcinogenic and non-carcinogenic endpoints.

There are several regulatory sources that report TRVs (i.e., RfDs, TDIs, MRLs, slope factors). Some of the most frequently used sources are Health Canada, the U.S. EPA Integrated Risk Information System (IRIS) database, the U.S. EPA National Center for Environmental Assessment (NCEA), the World Health Organization (WHO), and the Agency for Toxic Substances and Disease Registry

(ATSDR). Given that this assessment is within a Canadian jurisdiction, values provided by Health Canada were generally preferentially selected as TRVs for evaluation of the health impacts on human receptors. The Toxicology Evaluation Section of the Health Products and Food Branch of Health Canada has published TDIs for a number of trace elements found in foodstuff. These values were also considered for use in this assessment. The other above-mentioned sources were used to infill data gaps in the Health Canada database.

Table 6.4-1 provides a summary of the TRVs selected for use in the assessment for oral exposure. The TRVs, health effects (toxicological endpoints), and reference sources for each TRV are provided in the table. A discussion of the rationale for selecting the oral TRVs is provided in Attachment G.

Table 6.4-1 Selected Oral Toxicity Reference Values for Human Receptors

Constituent of Potential Concern	Effect Type [(a)]	Value	Health Effect	Reference [(b)]
Arsenic	C	1.8	Internal cancers (liver, lung, bladder, kidney) (oral, human)	Health Canada (2009)
	NC	0.001	Not provided	Health Canada Food Directorate (2002)
Cadmium	NC	0.001	Significant proteinuria	Health Canada (2009) IRIS (U.S. EPA 2011, last updated 1994)
Cobalt	NC	0.01	Haematological effects (polycythemia)	ATSDR (2004a)
Copper	NC	0.09	Hepatoxicity	Health Canada (2009)
Lead	NC	0.0036	Increase in blood lead concentrations in infants	Health Canada (2004c)
Molybdenum	NC	0.005	Increased uric acid levels	IRIS (U.S. EPA 2011, last updated 1993)
Nickel	NC	0.011	Decreased body and organ weights (oral exposure, rats)	Health Canada (2009)
Selenium	NC	0.005	Clinical selenosis	IRIS (U.S. EPA 2011, last updated 1991)
Uranium	NC	0.0006	Degenerative lesions in kidney tubules	Health Canada (2001)
Zinc	NC	0.3	Diminished copper status (required for protection against free radicals and oxidative stress)	IRIS (U.S. EPA 2011, last updated 2005)

Note: Units: mg/(kg-d) for all except for arsenic carcinogenic (C) TRVs which are in $(mg/(kg\text{-}d))^{-1}$

a = C – Carcinogenic (non-threshold) effect; NC - non-carcinogenic (threshold) effect

b = IRIS - Integrated Risk Information System on-line database (last updated represents year of last significant revision)

ATSDR = Agency for Toxic Substances and Disease Registry

6.4.2 Health-Based Values for Gaseous Pollutants and Dust

In general, the adverse effects of exposure to gaseous air pollutants are associated with irritation of the tissues of the eyes and upper and lower respiratory systems. Exposures to the gaseous air pollutants (i.e., SO_2, NO_2) are assessed using air quality guidelines (AQG) values obtained from the WHO that are designed to offer guidance in reducing the health impacts of air pollution. The NO_2 guidelines are health-based values with no safety factors built in and are not targets, while the SO_2 values (Tier I and II) are only targets whereas the guideline is health-based. However, since no other TRVs are presently available to evaluate the health effects relating to NO_2 and SO_2 exposure, and since the WHO values are the most current health-based values available, they were used as health-based values in the assessment. Table 6.4-2 provides a summary of the health-based values used for the assessment of gaseous air pollutants.

Table 6.4-2 Health-Based Values for Gaseous Air Pollutants

Gaseous Air Pollutants		Health-Based Concentration (µg/m^3)	Jurisdiction
NO_2	1 hour	200	WHO (2005)
	Annual	40	WHO (2005)
SO_2	1 hour	500	WHO (2005)
	24 hour	20*	WHO (2005)

Note:

* Although the WHO has set the guideline at 20 µg/m^3, it is acknowledged that there will be difficulty in achieving this guideline. As such, a stepped approach has been suggested, using a tier I interim guideline value of 125 µg/m^3 and a tier II interim value of 50 µg/m^3.

There is a growing body of scientific studies linking air pollutants to health effects. Recent assessments of the available health data are implying a stronger link between particulate matter (PM) and health effects resulting from short- and long-term exposures. In addition, the effects are estimated to occur at levels that are lower than previously believed. This has motivated some regulators to re-assess the potential impact of particulate matter pollution on public health (CARB 2008).

For particulate matter 10 micrometers or less (PM_{10}), the reference level of 25 µg/m^3, based on a 24 hour averaging time which resulted in consistent associations observed in epidemiological studies with mortality and hospitalizations as well as concerns over links to chronic bronchitis and cardiovascular disease (CEPA/FPAC WGAQOG 1998), was selected as a health-based limit.

A threshold value range of 5 to 7 µg/m^3 (CARB 2008) was used as the health-based level for particulate matter 2.5 micrometers or less ($PM_{2.5}$) (which encompasses ultrafine particles) and the epidemiological evidence related to short-term exposures are the most relevant to consider in this assessment. This threshold range was also considered in this assessment as a health-based limit.

6.5 Radiological Assessment Criteria

The assessment of effects from exposure to radioactive constituents involves estimation of the combined (total) dose that a VEC may receive from radionuclides taken into the body as well as from exposure to radiation fields in the external environment. In addition, it is standard practice to take into account differences in the effects of alpha, beta and gamma radiation. These factors are discussed below.

6.5.1 Relative Biological Effectiveness Factors

Radiation effects on biota depend not only on the absorbed dose, but also on the relative biological effectiveness (RBE) of the particular radiation (i.e., alpha, beta or gamma radiation). For example, alpha particles can produce observable damage at lower absorbed doses than gamma radiation. Thus, in order to estimate the potential harm to non-human biota from a given absorbed dose, the absorbed dose is multiplied by an appropriate radiation weighting factor. This in turn is derived from experimentally determined RBE. In this assessment, the terms “RBE” and “radiation weighting factor” are used interchangeably.

For the purposes of human radiological protection, each component of the absorbed dose to a tissue or organ is weighted according to the radiation quality. The appropriate RBE value for biota is the subject of ongoing scientific discussion. Although an RBE of 20 is used for human radiation protection, according to the review by UNSCEAR (1996):

“In the case of wild organisms, however, it is likely to be deterministic effects that are of greatest significance, and for alpha radiation the experimental data for animals indicate that a lower weighting factor, perhaps 5, would be more appropriate; the weighting factors for beta and gamma radiation would remain unity.” (para. 18)

A review of the literature by Trivedi and Gentner (2000) concluded that:

“since the majority of studies report RBE values less than or equal to 10 for endpoints, and doses and dose rates that are more ecologically significant, a value of 10 might be appropriate for weighting doses to evaluate the effects of alpha emitters at the population level, if any”.

Also, the U.S. DOE (2000) reviewed the issue and recognized that the critical biological endpoint of concern in radiation exposures of biota appears to be deterministic, and that the radiation weighting factor for deterministic effects is substantially less than the corresponding average quality factor used in radiation protection of humans (i.e., 20). Based on this information, U.S. DOE concluded that the radiation weighting factor for deterministic effects appears to lie in the range of about 5 to 10.

However, as interim guidance, they recommend the use of an RBE of 20 in the proposed standard (U.S. DOE 2000).

A wide range of RBE values for internally deposited alpha particles has been published. The PSL2 assessment (EC/HC 2003) suggests an RBE of 40. A report of the (former) Advisory Committee on Radiological Protection (ACRP 2002) suggested a nominal RBE value of 10 with a range of about 5 to 20 for non-human biota (ACRP 2002). A recent report of the European Community suggests using an RBE of 10 to illustrate the effect of alpha RBE. For the purposes of this assessment, uncertainty associated with the choices of RBE is acknowledged and a RBE value of 10 was used, consistent with N288.6 (CSA 2012).

6.5.2 Reference Dose Levels for Aquatic Biota

The Canadian Standards Association (CSA) suggests a dose rate of 9.6 milliGrays per day (mGy/d) as the reference dose level below which population effects to aquatic biota would not be expected (CSA 2012). This value is based on UNSCEAR (2008).

The reference dose rates for aquatic biota that were used in the assessment are summarized in Table 6.5-1.

Table 6.5-1 Reference Dose Rates for Aquatic Organisms

Aquatic Organism	Reference Dose Rate
Aquatic plants (algae and macrophytes)	9.6 mGy/d
Benthic invertebrates	9.6 mGy/d
Fish	9.6 mGy/d
Zooplankton	9.6 mGy/d
Phytoplankton	9.6 mGy/d
Note: mGy/d = milliGray per day Reference dose rates based on N288.6 (CSA 2012)	

6.5.3 Reference Dose Levels for Terrestrial Biota

The IAEA (1992) published the results of an assessment of the effects of acute and chronic radiation on terrestrial populations and communities. They reached several general conclusions regarding chronic radiation: reproduction is likely to be the most limiting endpoint in terms of population maintenance, and irradiation at chronic dose rates of 1 mGy/d or less does not appear likely to cause observable changes in terrestrial animal populations. Also, they concluded that irradiation at chronic dose rates of 10 mGy/d or less does not appear likely to cause observable changes in terrestrial plant populations.

The CNSC has provided a reference dose rate of 3 mGy/d as an appropriate limit for small mammals and terrestrial plants (Bird et al. 2002; EC/HC 2003). This reference dose rate is based on reproductive endpoints for small mammals. In the absence of data for avian species, the CNSC suggest that the reference dose rate for small mammals should also apply. These guidelines were also considered in the assessment. N288.6 (CSA 2012) recommends a radiation dose rate limit folloing UNSCEAR (2008) of 2.7mGy/d for terrestrial biota. For consistency with N288.6, the reference dose rate for terrestrial biota (birds and mammals) used in the assessment is 2.7 mGy/d.

6.5.4 Reference Dose Levels for Human Health

Assessment of radiation exposures to members of the public is commonly based on estimation of the incremental effects of a project or site. Such assessments consider the radiation dose received from direct exposure to gamma radiation as well as the dose received from ingestion of radionuclides. The human model (described in Section 2.10) converts radionuclide intake by the human receptors from the various pathways into a dose. Potential effects from radiation are compared to an incremental dose limit of 1,000 microSieverts per year (µSv/y) (1 milliSievert per year [mSv/y]) recommended by the Canadian Nuclear Safety Commission (CNSC) for the protection of members of the public. In addition, the dose constraint of 300 µSv/y (0.3 mSv/y) recommended by Health Canada in the Canadian NORM Guidelines (Health Canada 2000) is considered. Doses below this level are considered as “unrestricted” and no further action is needed to control doses or materials.

7 Results for the Receiving Environment

Predicted air quality for the assessment area is presented for the evaluation of potential effects. Predicted water and sediment quality for Judge Sissons Lake for the O1Ex22 effluent release scenario is presented for the evaluation of potential effects on the aquatic environment. The O1Ex22 effluent release scenario was determined previously (Section 3.4.2) to be a conservative estimate of the potential effects associated with the discharge, operation, and closure options proposed. Potential effects on downstream water quality are also discussed.

7.1 Predicted Air Quality

The air modelling documents (Appendix 4B Air Quality) present the detailed results of predicted air quality for dust, NO_2, SO_2, and metals. The predictions were made for phased site operations as well as a maximum emission scenario.

Appendix 4B Air Quality compared the predicted air concentrations to air quality criteria within the atmospheric environment assessment. The estimated concentrations of dust, NO_2 and SO_2 are used directly in the assessment of human health and for ecological receptors and this information is presented in the relevant sections of this assessment.

As outlined in Appendix 4B Air Quality, all predicted maximum 24-hour metal concentrations are well below applicable criteria (Table 7.1-1). In contrast, predicted maximum 24-hour uranium concentrations are above the recently promulgated Ontario Ambient Air Quality Criterion (AAQC) of 0.3 µg/m^3. As a result a frequency analysis was completed. At the Accommodation Complex, the 24-hour uranium AAQC is only exceeded once. The frequency analysis also indicated that the number of exceedances within about 1 km of the Kiggavik site are limited to one day. Similarly, within about 500 m or less of the Sissons site, exceedances are also limited to one day.

The air concentrations from the air dispersion modeling for metals are used in the pathways assessment. The activity of U-238 was determined based on the specific activity of uranium (0.01235 Bq U-238/µg of natural uranium); the levels of the other radionuclides in the uranium-series (Th-230, Ra-226, Pb-210 and Po-210) were taken to be equal to U-238. The air concentrations were used to determine the potential impact on soil and vegetation as well as inhalation by people.

Table 7.1-1 Incremental Maximum 24-hr Concentrations of Metals at Discrete Receptor Locations

Receptor Name	24-hour Maximum Concentration (µg/m^3)									
	U	As	Cd	Co	Cu	Pb	Mo	Ni	Se	Zn
Accommodation Complex	3.1E-01	8.1E-04	5.3E-05	3.6E-03	6.1E-03	1.8E-02	7.6E-03	1.1E-02	4.9E-04	1.1E-02
Community of Baker Lake	9.7E-04	8.3E-06	2.7E-07	1.5E-05	2.8E-05	5.6E-05	1.8E-05	6.0E-05	2.9E-06	4.6E-05
Judge Sissons Lake Cabin	1.7E-02	1.6E-04	5.7E-06	2.4E-04	5.0E-04	9.7E-04	3.7E-04	1.3E-03	6.0E-05	7.1E-04
Air Quality Criteria (µg/m^3)	0.3	0.3	0.025	0.10	50	0.50	120	0.2*	10	120

Notes:

Bold shaded text indicates that a value is greater than the air quality criteria. Only one (1) of exceedance was predicted at the Accommodation Complex.

24-hour criteria provided in Ontario's Ambient Air Quality Criteria, Ontario Ministry of Environment Standards and Development Branch, 2008.

*Nickel AQ criteria adopted from the Ontario Ministry of Environment's new standard under Regulation 419/05 accepted June 22, 2011. EBR No. 010-7190.

Uranium AQ criteria adopted from the Ontario Ministry of Environment's new AAQC under Regulation 419/05 accepted June 22, 2011. EBR No. 010-7190.

The Kiggavik project will release both uranium-bearing dust and radon. Therefore, Pb-210 will also be present in the air from the decay of radon released from the operations. The presence of Pb-210 that is a result of the decay of Rn-222 is not considered in the air dispersion modelling. The ingrowth of Pb-210 from the radon depends on the "age" of the radon at any location/receptor, which in turn depends on the wind speed and distance of the receptor from the radon source. The greater the distance, the greater the ingrowth of Pb-210. For present purposes, receptors at varying distances from the Kiggavik project were considered; both Kiggavik and Sissons operations were assumed to be active. The furthest receptor considered was Baker Lake which is approximately 90 km from the project. At shorter distances, the ingrowth of Pb-210 would be expected to be less. Assuming conservatively that the radon decays directly to Pb-210 (the short-lived radon decay products with effective half-lives of about 30 minutes were neglected), the ingrowth of Pb-210 from radon is approximately given by (based on Evans (1955, Chapter 15, equation 2.8 for radioactive series decay):

$$Pb-210\ (\frac{Bq}{m^3}) = C_{Rn} \times \left[\frac{\lambda_b}{\lambda_b - \lambda_a}\right] \times (e^{-\lambda_a T} - e^{-\lambda_b T}) \qquad (7.1\text{-}1)$$

Where:

C_{Rn} Radon concentration in air (Bq/m^3)

λ_a Decay constant of Rn-222, 2.098×10^{-6} s^{-1} [ln(2) / half-life of Rn-222]

λ_b Decay constant of Pb-210, 9.901×10^{-10} s^{-1} [ln(2) / half-life of Pb-210]

T Transit time from source to location

Using the above equation, the Pb-210 in air produced from the decay of radon was calculated and compared to the Pb-210 concentration in uranium dust at the same locations. The results are shown in Table 7.1-2.

Table 7.1-2 Estimate of Pb-210 Levels in Air Consideration Contribution from Radon

Receptor	U dust concentrationa (µg/m3)	Radon concentrationa (Bq/m3)	Distance from Kiggavik sources (km)	Distance from Sissons sources (km)	Travel Timeb (s)	Pb-210 in dustc (Bq/m3)	Pb-210 from Rnd (Bq/m3)
Kiggavik Camp	1.7E-02	1.2E+01	1	17.5	2.4E+02	2.1E-04	3.0E-06
Judge Sissons Lake	6.3E-04	2.9E-01	10	12.5	2.7E+03	7.8E-06	7.8E-07
Baker Lake	1.7E-05	8.2E-03	85	95	2.2E+04	2.1E-07	1.7E-07

Notes:

a Annual average concentration from Scenario 2

b. Based on average wind speed of 4.1 m/s and average distance to sources, except for 1 km used for Kiggavik Camp sources.

c. Based on 1 µg uranium = 0.01235 Bq U-238 = Pb-210.

d. Based on Evans (1955) - see text

The uranium and dust concentrations used were the predicted annual concentrations when both sources (Kiggavik and Sissons) were assumed to be in operation. As shown in the table, the Pb-210 concentration from the decay of radon is a small fraction (10% or less) of the Pb-210 concentration in the dust out to distances of about 10 km from the sources. At Baker Lake, an average of 90 km from the Kiggavik and Sissons operations, the ingrown Pb-210 is almost equal (84%) to the Pb-210 in the airborne dust. However, at such a distance, the radon concentration, and corresponding Pb 210, is insignificant and immeasurable relative to typical background concentrations. Therefore, Pb-210 resulting from the decay of Rn is neglected.

7.2 Predicted Water Quality

Figure 7.2-1 presents the monthly predicted water concentrations in the eight segments of Judge Sissons Lake for the O1Ex22 effluent release scenario. The four phases of effluent release are indicated on the figures: baseline conditions (pre-operation), operation phase, final closure phase, and post-closure phase.

The baseline results provide the water quality in each segment under natural conditions and the impact of the winter ice cover is evident in this phase of the simulation. Comparison of the range of concentrations in the baseline phase among the various Judge Sissons Lake segments illustrates the effect of the ice cover on the different segments. For example, JSL-1 is shallow compared with JSL-4 and, therefore, there is much fluctuation between the summer and winter concentrations in JSL-1 (for uranium, from 0.1 to 0.22 µg/L) compared with concentrations in JSL-4 (for uranium, from 0.1 to 0.12 µg/L).

The operation phase shows the predicted water quality results under the assumed effluent release for the extended operation of Option 1, which considers discharge of the Kiggavik WTP to JSL-2 and Sissons WTP to JSL-8. The effluent is distributed throughout Judge Sissons Lake by natural flow processes and horizontal dispersion. Figure 7.2-1 illustrates the range of predicted water concentrations in each of the Judge Sissons Lake segments during the operation phase. Comparison of the baseline phase with the operation phase gives an indication of which COPC in the effluent have more of an influence on the water quality in the various Judge Sissons Lake segments. For example, comparison between the baseline and operation phase concentrations of arsenic at JSL-1 shows that arsenic in the effluent discharge creates a noticeable difference, while concentrations of thorium-230 are essentially the same between the two phases.

The final closure phase represents the predicted water quality under the bounding case decommissioning scenario with 22-years of Main Zone consolidation. As effluent releases are gradually decreased through this phase, Figure 7.2-1 shows that water quality is predicted to approach baseline concentrations as effluent discharge ceases. The system is shown to respond relatively quickly to changes in the effluent release scenario.

Finally, the post-closure phase shows the recovery of predicted water quality for Judge Sissons Lake after effluent releases have stopped. Again, the system responds quickly and returns to baseline/pre-operation conditions, and the water effluent scenario assessed indicates that the Project does not have long-term impacts on the water quality of Judge Sissons Lake.

The associated water quality guidelines (Section 6.1) are indicated on the graphs. Since the peak concentrations are of interest in this assessment and these occur during the winter, the WQG shown are those for the mean hardness expected during winter conditions. For the COPC with water quality guidelines available, predicted water concentrations of uranium, thorium-230, lead-210, radium-226,

polonium-210, arsenic, cobalt, lead, molybdenum, nickel, un-ionized ammonia, chloride, and sulphate, were below the guidelines in all segments of Judge Sissons Lake. For TDS, the predicted water concentrations in Judge Sissons Lake are below the WQG of 500 mg/L, the 95th percentile concentrations is at the WQG at times during the Operation Phase.

For cadmium, predicted concentrations in water exceed the Water Quality Guideline (WQG) at all points of the assessment in most segments of Judge Sissons Lake.

The updated 2014 CCME CWQG for cadmium is related to hardness. As discussed previously, due to the presence of the ice cover in Judge Sissons Lake there is expected to be fluctuation in concentrations during the year. This was examined for JSL-1 which is a shallow segment directly adjacent to the Kiggavik WTP discharge location and JSL-4 a larger segment that represents the outflow from the lake. Table 7.2-1 provides the estimated hardness; the calculated hardness-specific criterion from CCME; and, the estimated cadmium concentration for JSL-1 and JSL-4 separately for the summer and winter periods. This table shows that cadium concentrations, both at the mean and 95th percentile, remain below the WQG during the summer period. During the winter, the mean concentration is within the WQG but the 95th percentile does exceed the WQG. As summer is the period of growth and reproduction, this is the period of most relevance.

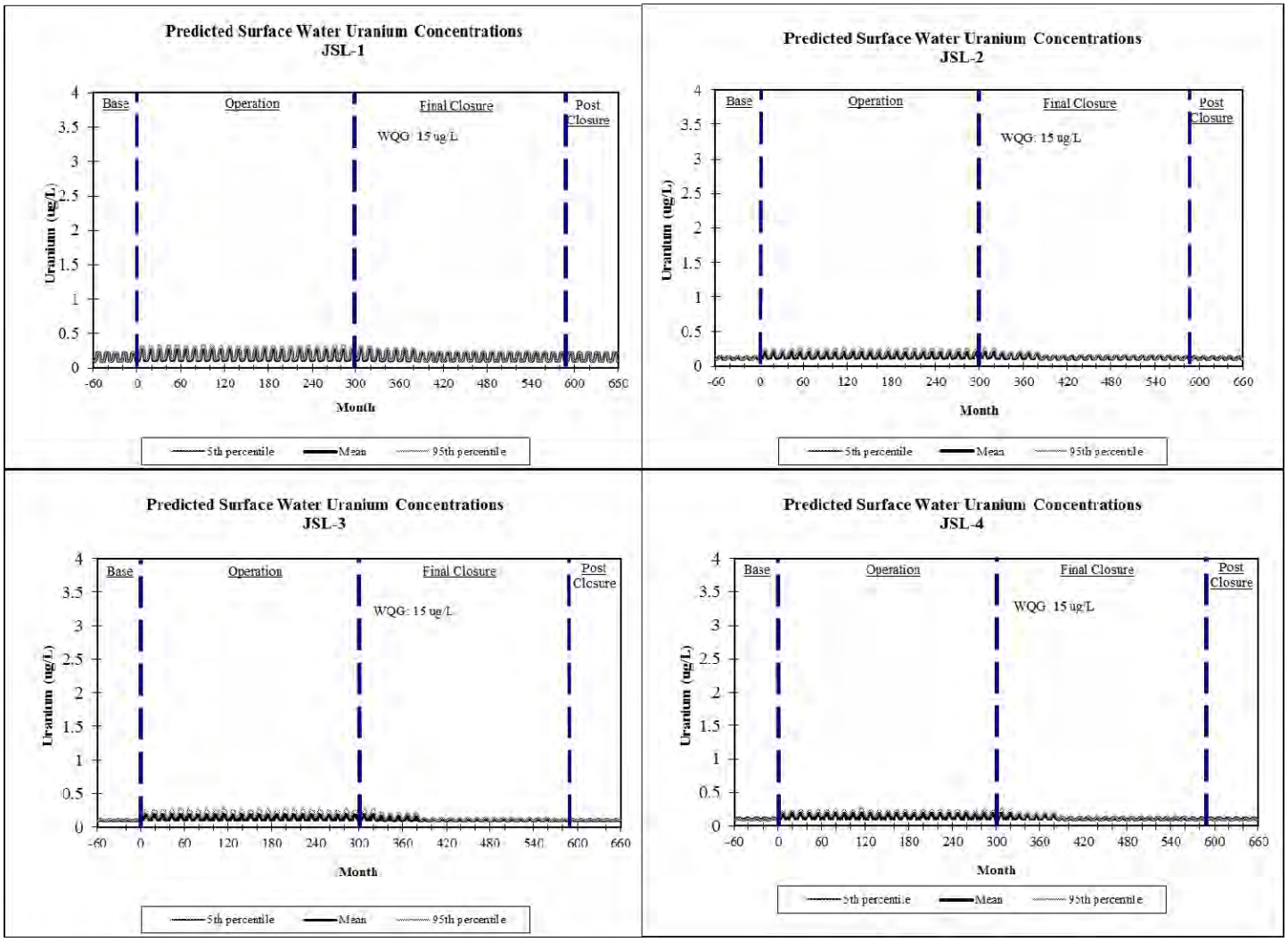


Figure 7.2-1 Water Quality Predictions

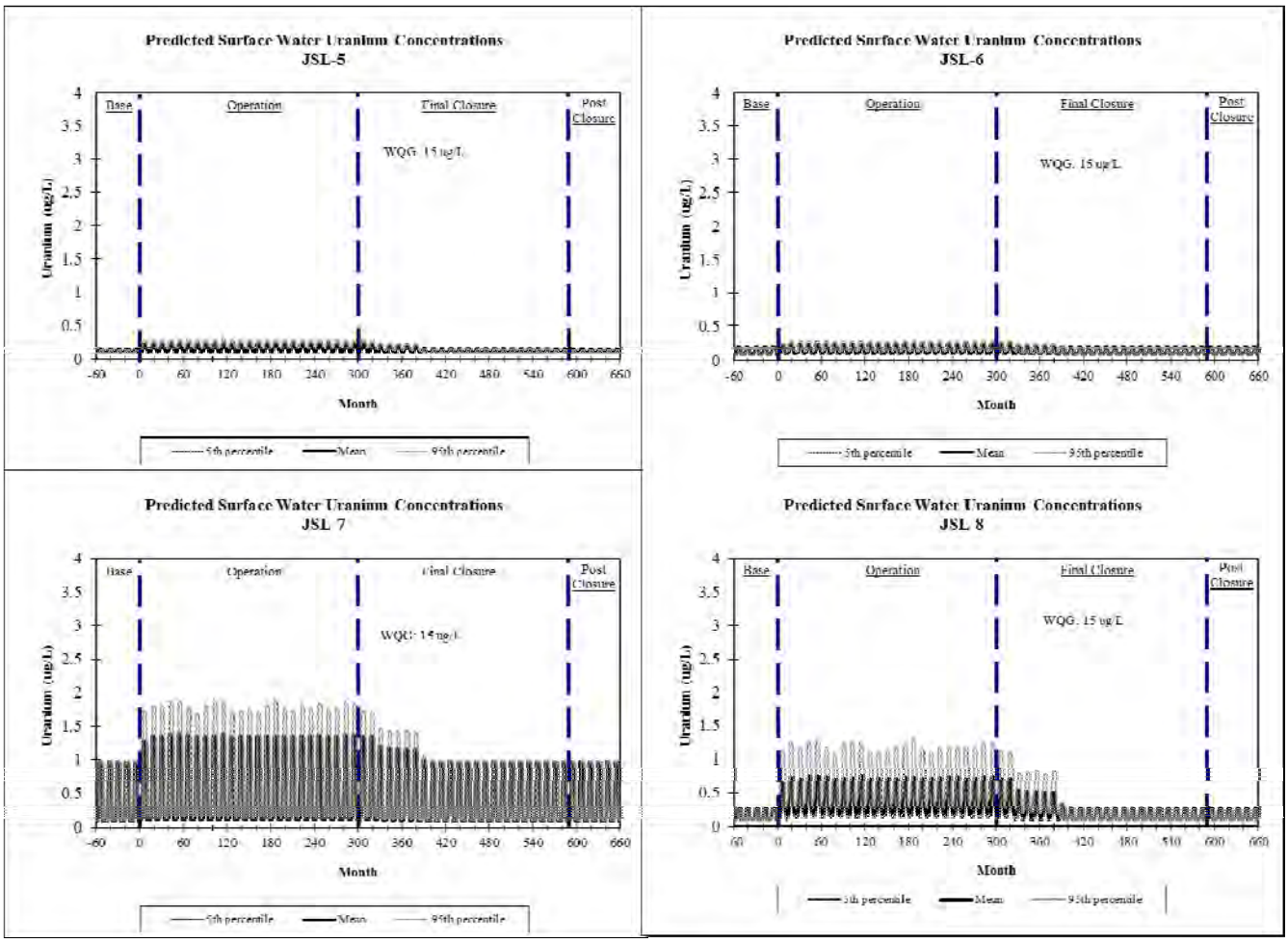


Figure 7.2-1 Water Quality Predictions (Cont'd)

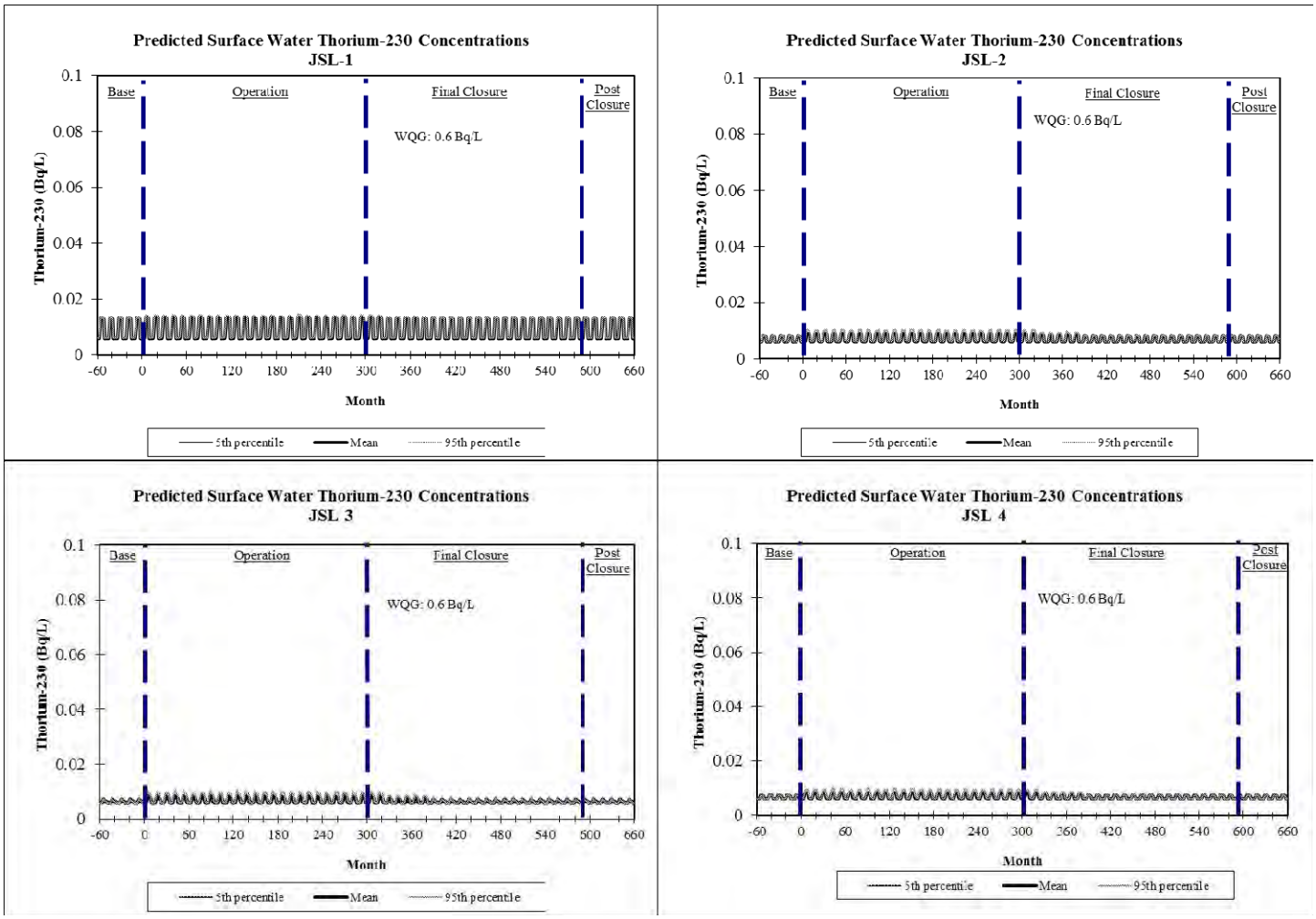


Figure 7.2-1 Water Quality Predictions (Cont'd)

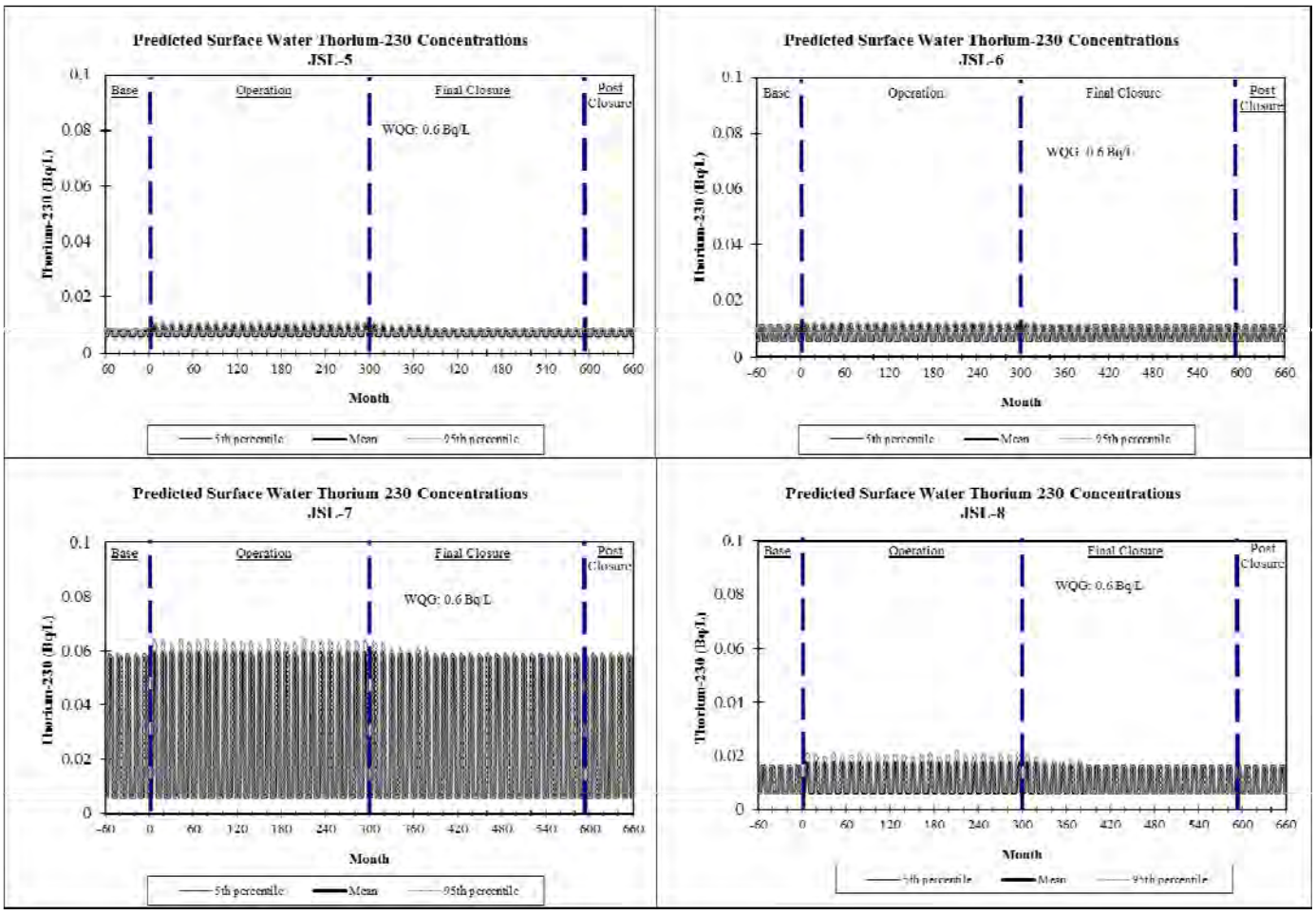


Figure 7.2-1 Water Quality Predictions (Cont'd)

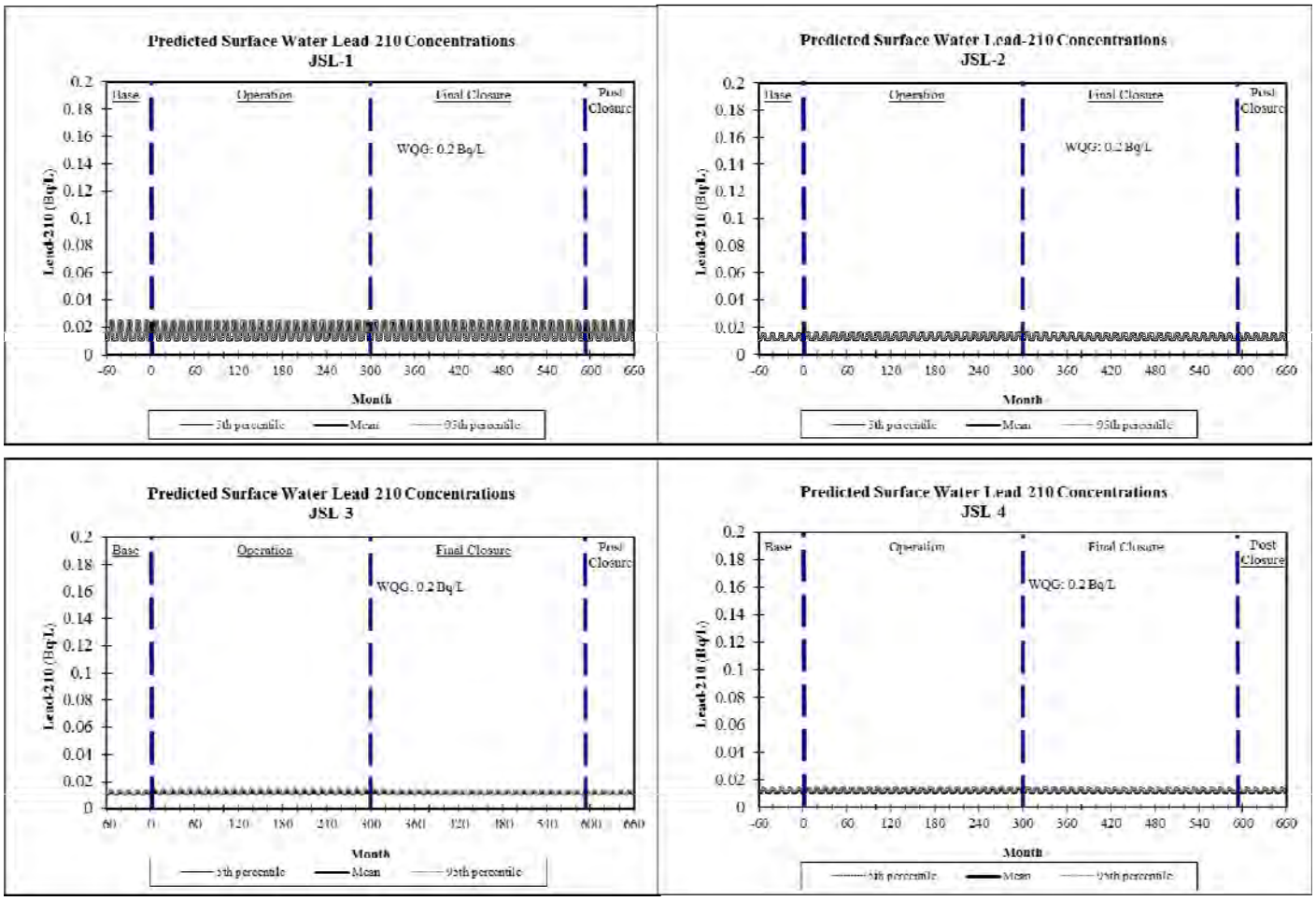


Figure 7.2-1 Water Quality Predictions (Cont'd)

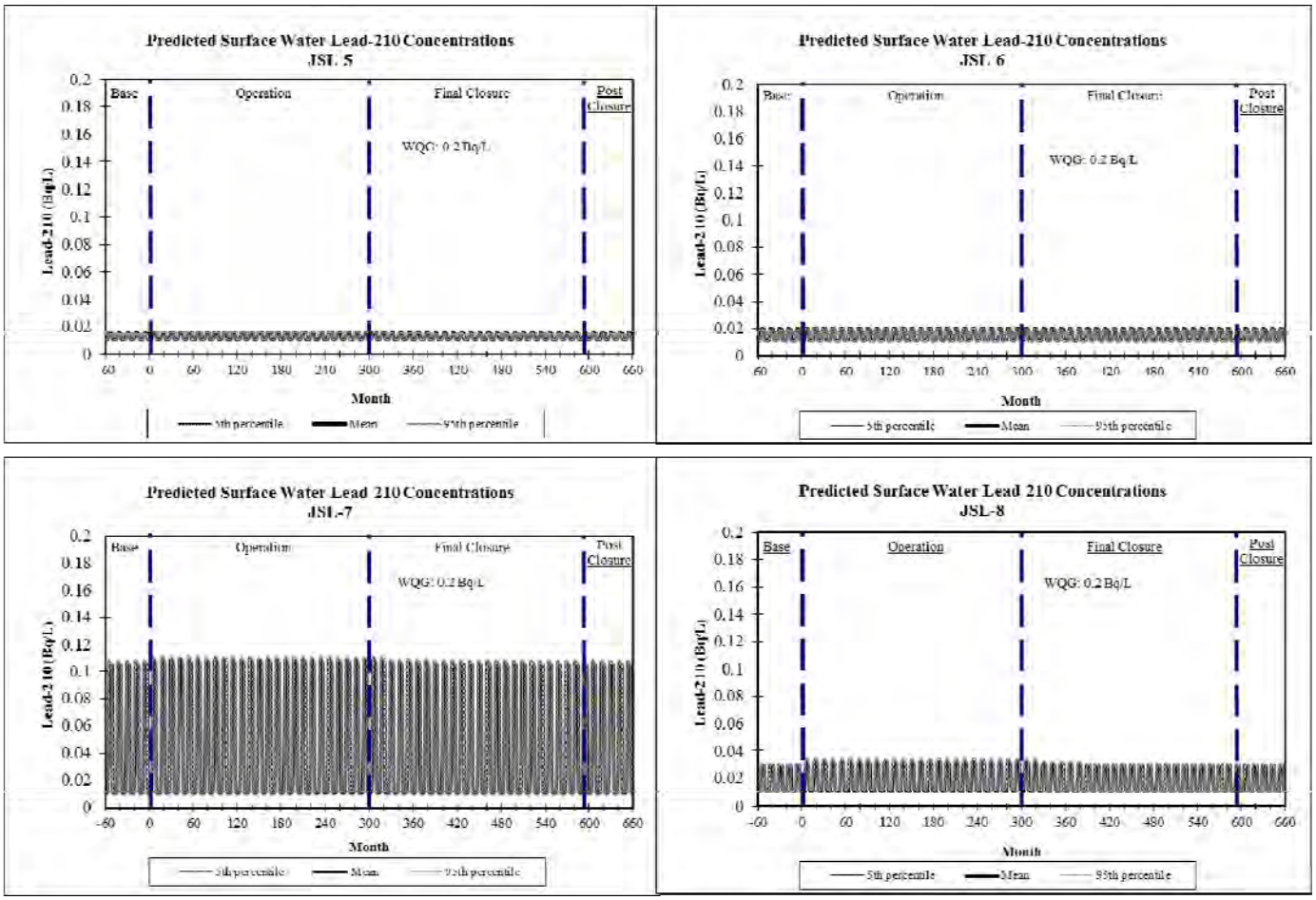


Figure 7.2-1 Water Quality Predictions (Cont'd)

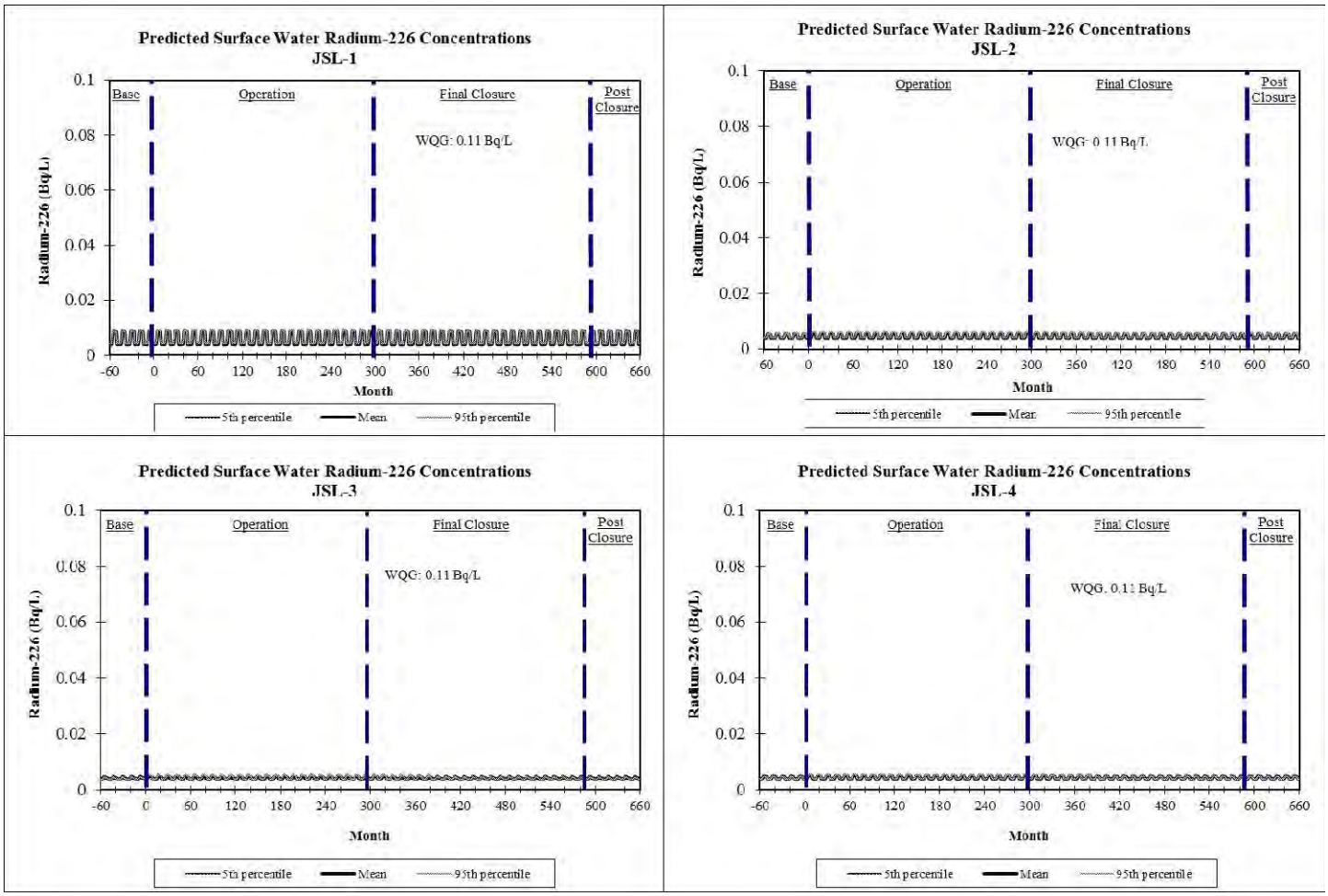


Figure 7.2-1 Water Quality Predictions (Cont'd)

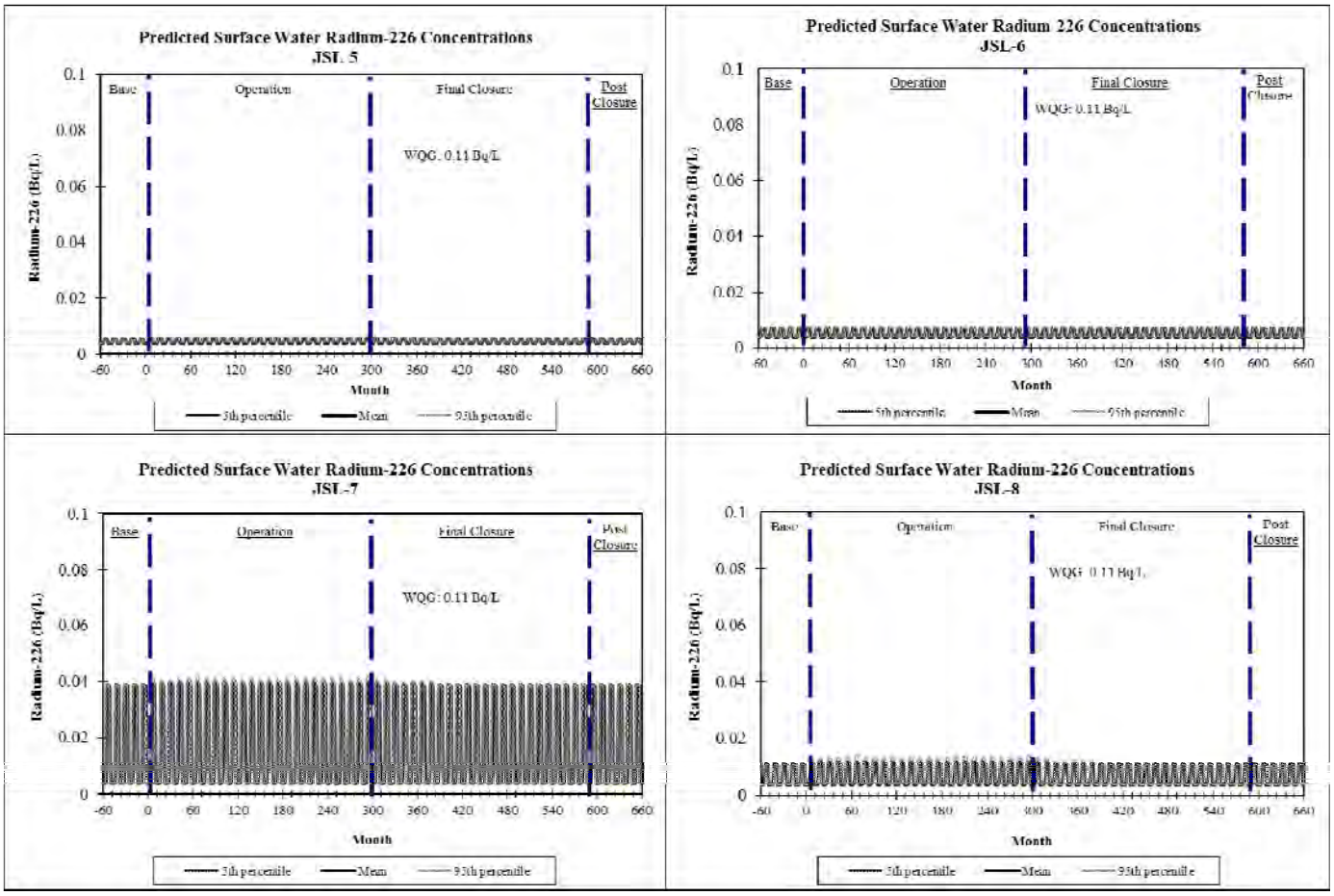


Figure 7.2-1 Water Quality Predictions (Cont'd)

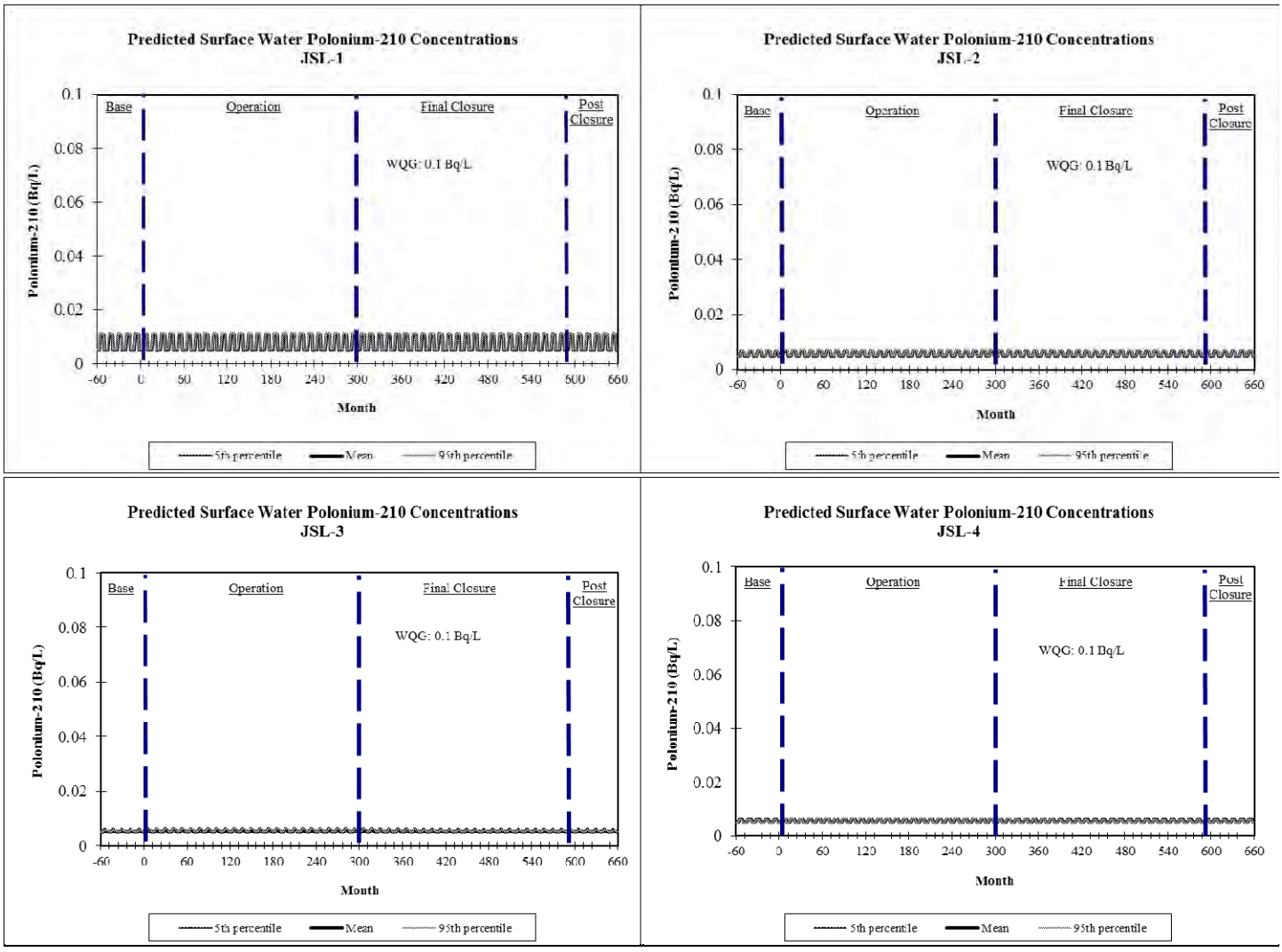


Figure 7.2-1 Water Quality Predictions (Cont'd)

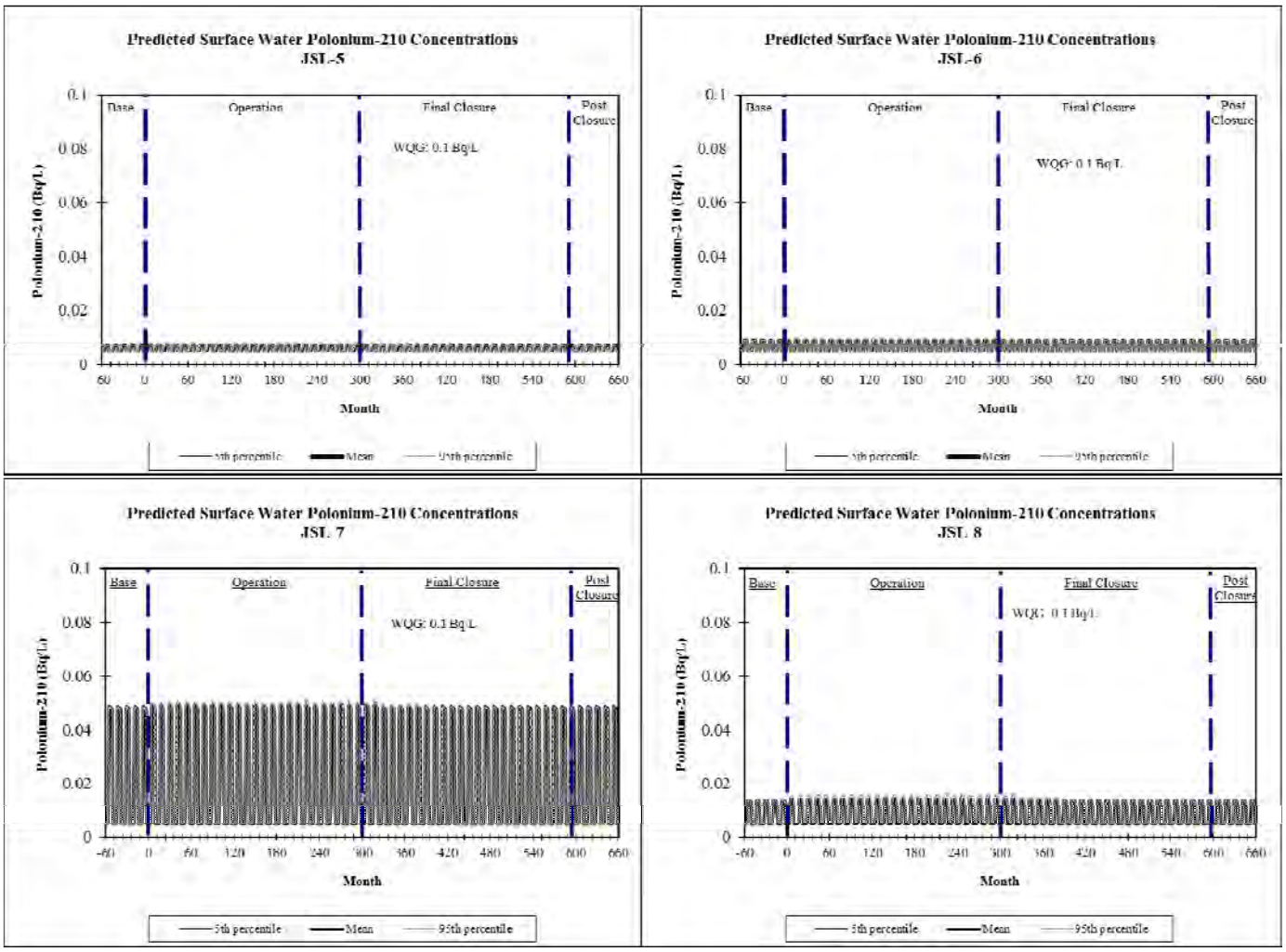


Figure 7.2-1 Water Quality Predictions (Cont'd)

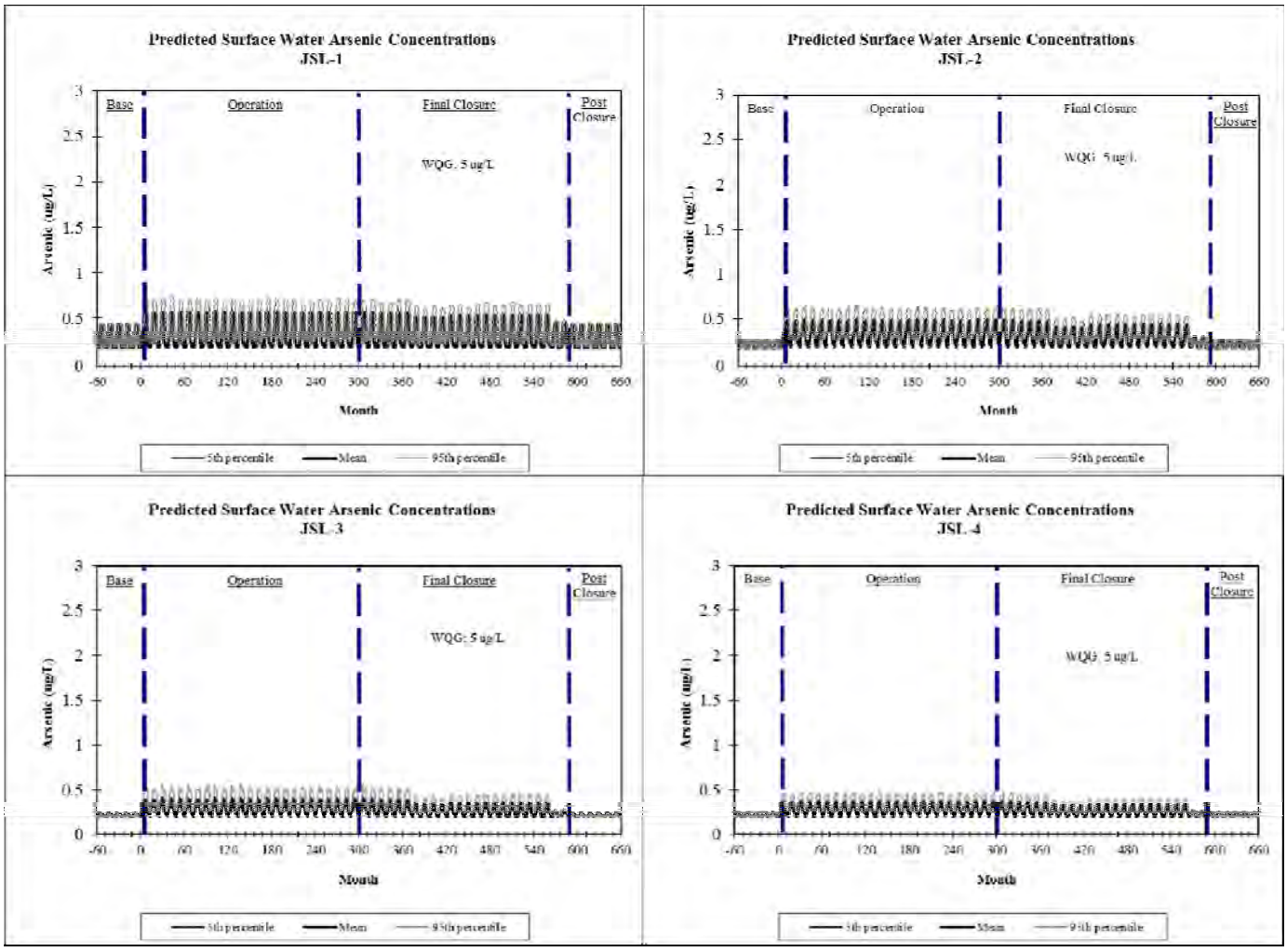


Figure 7.2-1 Water Quality Predictions (Cont'd)

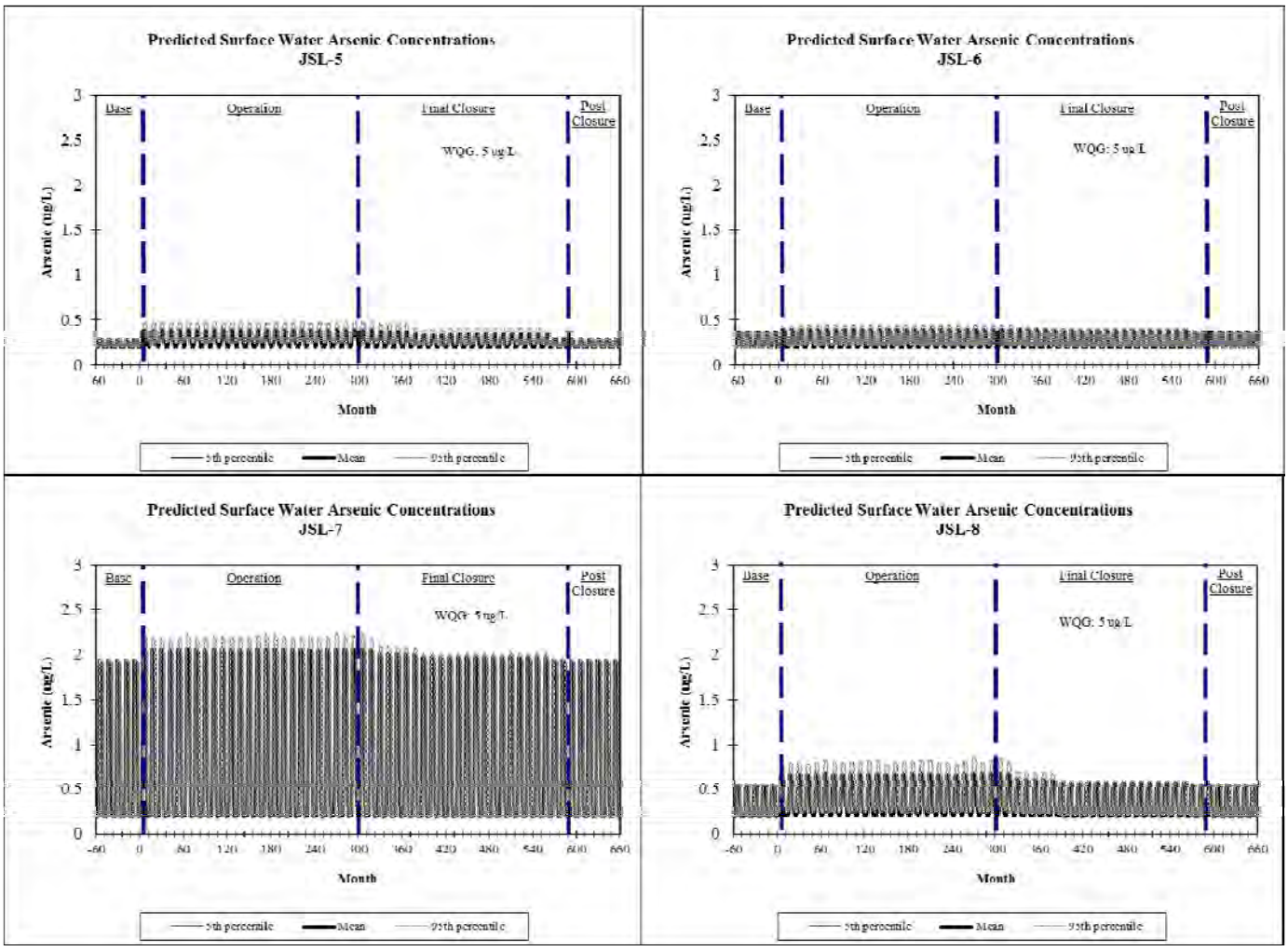


Figure 7.2-1 Water Quality Predictions (Cont'd)

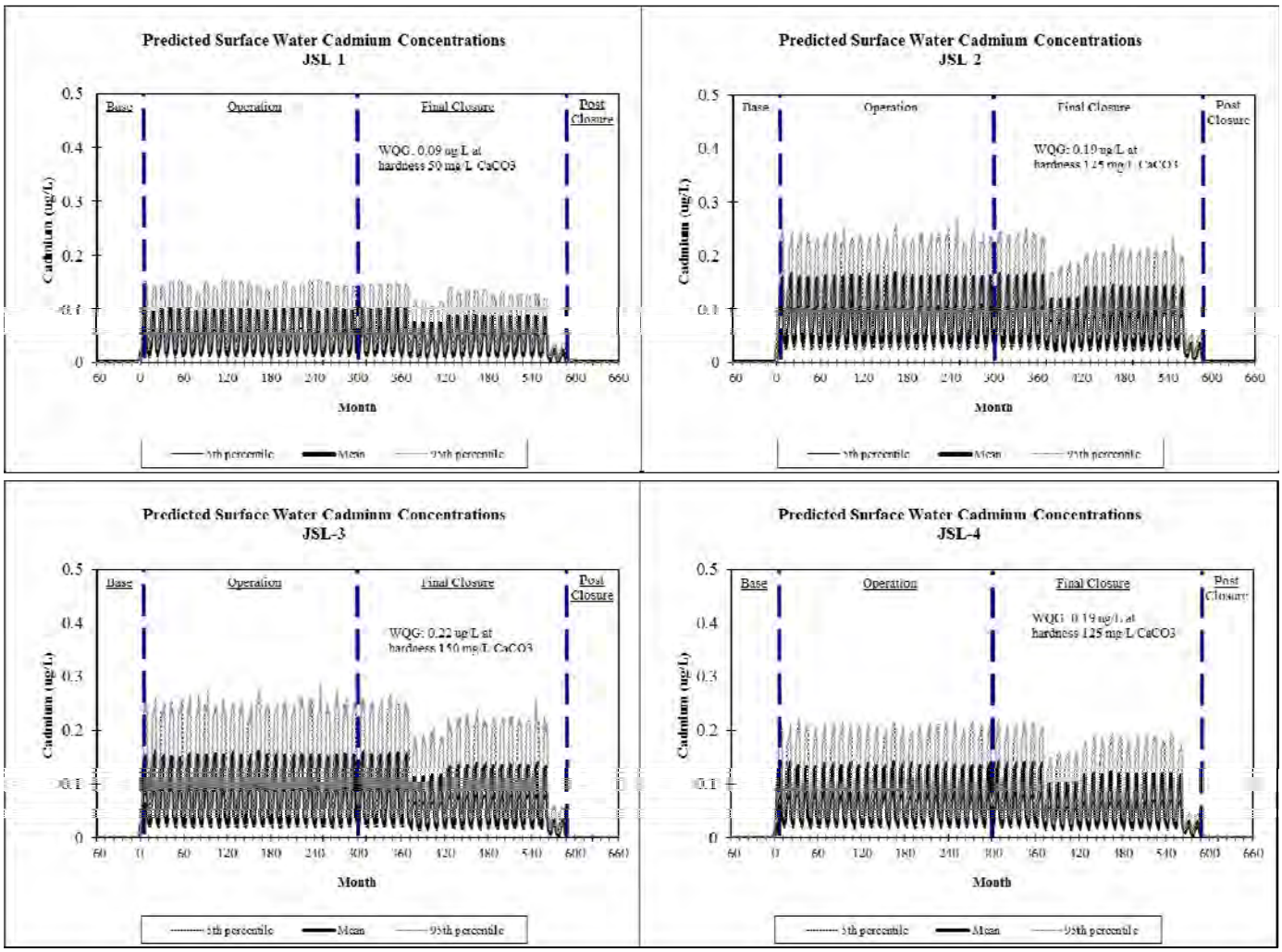


Figure 7.2-1 Water Quality Predictions (Cont'd)

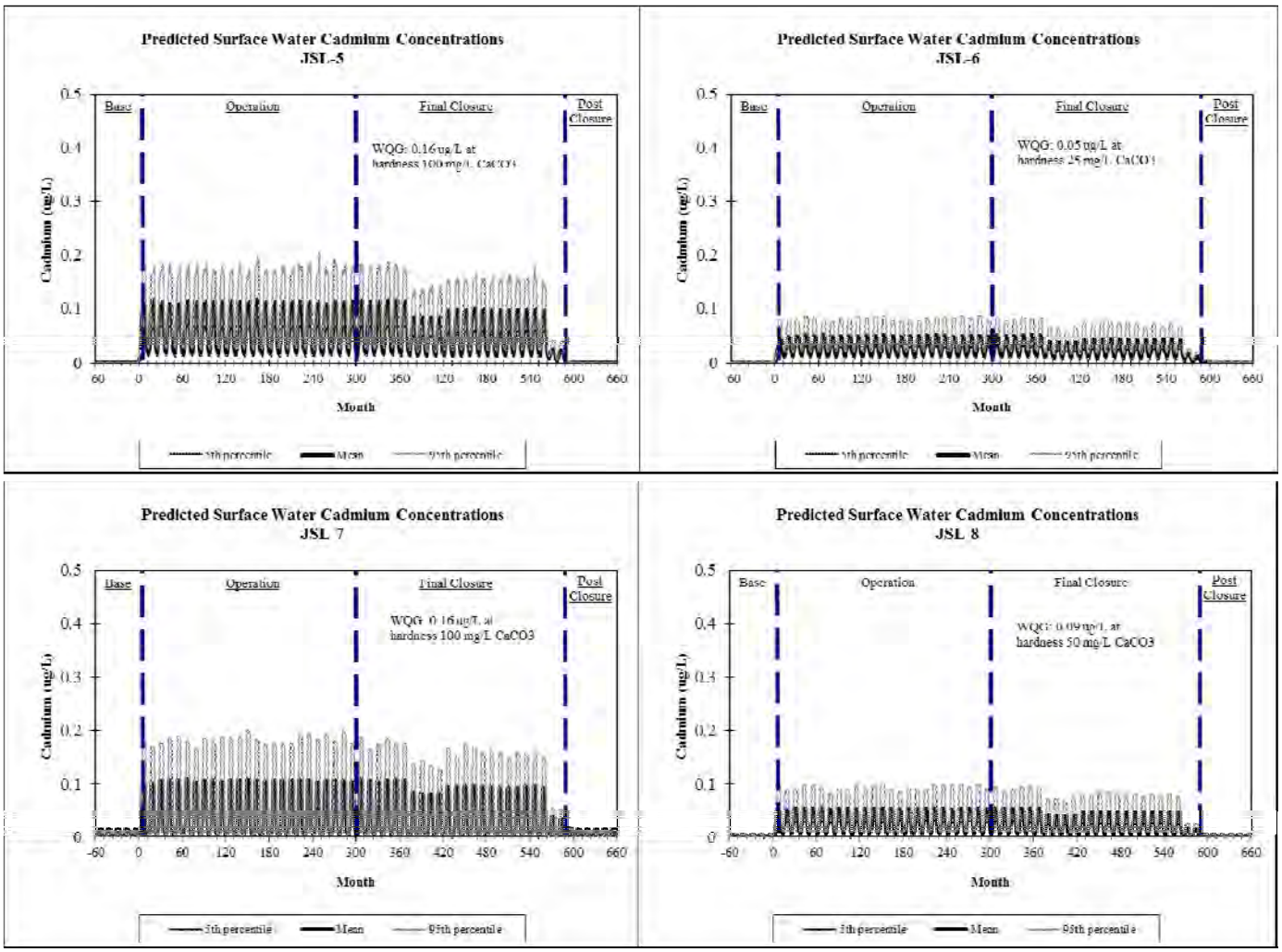


Figure 7.2-1 Water Quality Predictions (Cont'd)

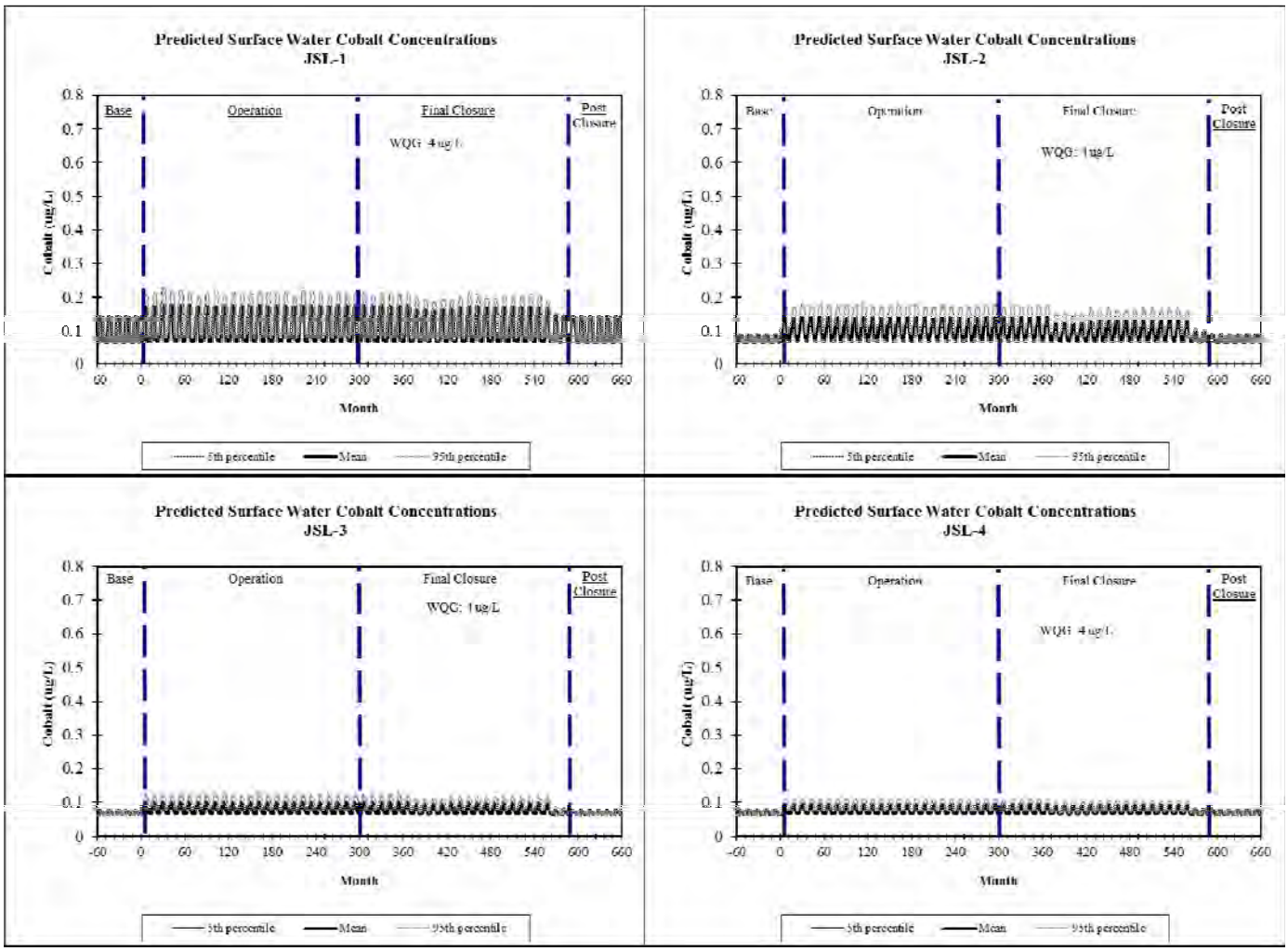


Figure 7.2-1 Water Quality Predictions (Cont'd)

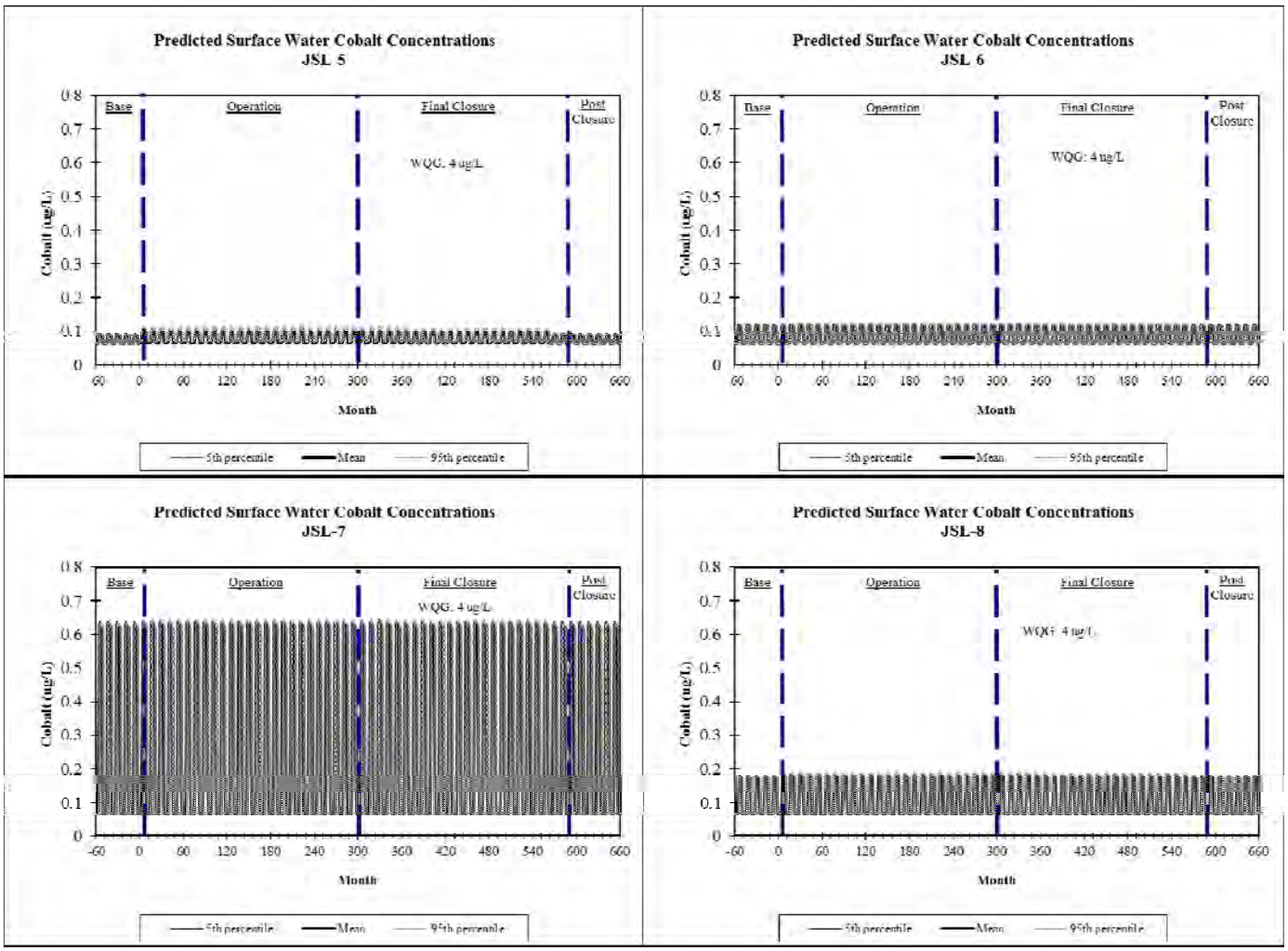


Figure 7.2-1 Water Quality Predictions (Cont'd)

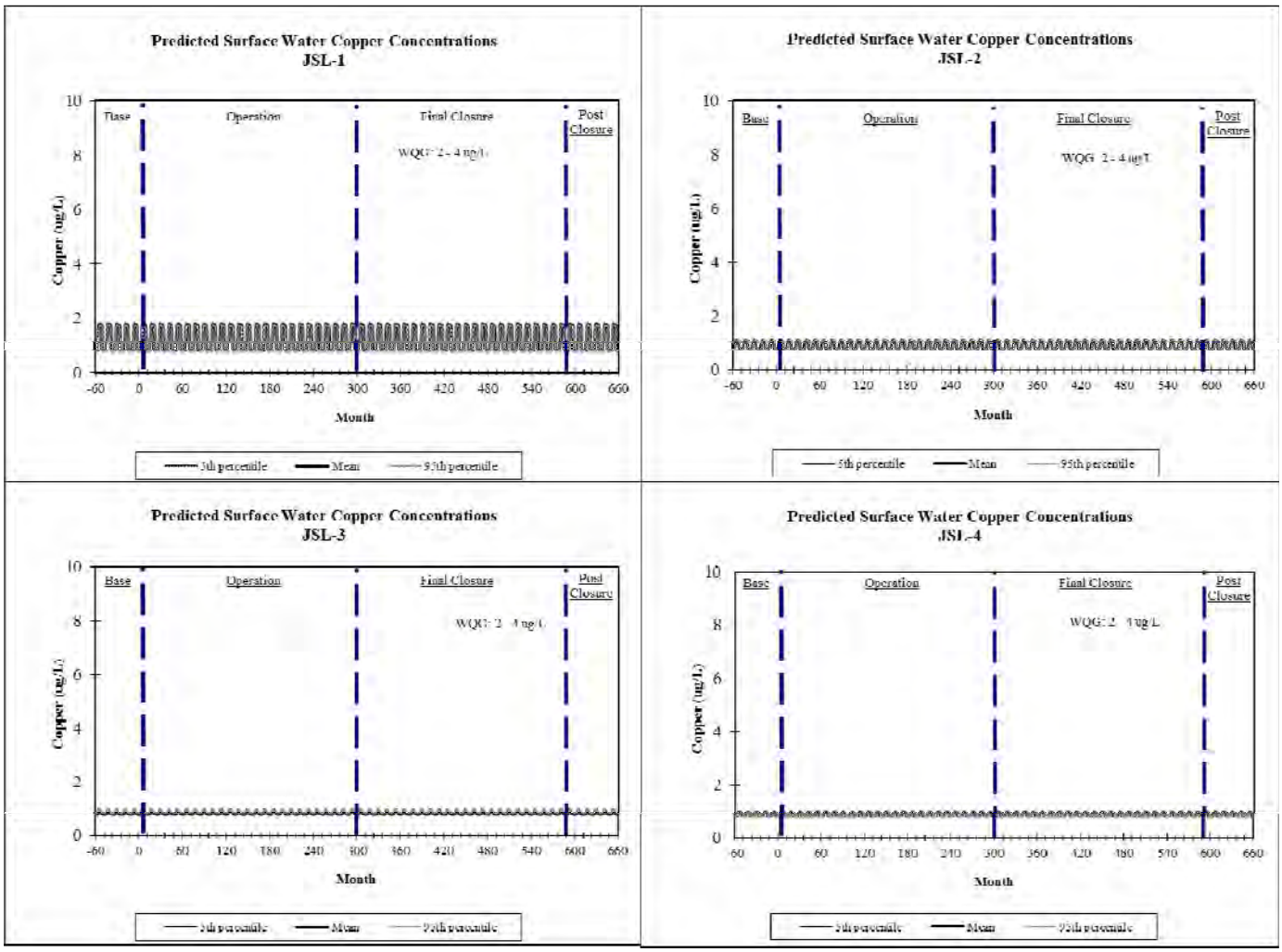


Figure 7.2-1 Water Quality Predictions (Cont'd)

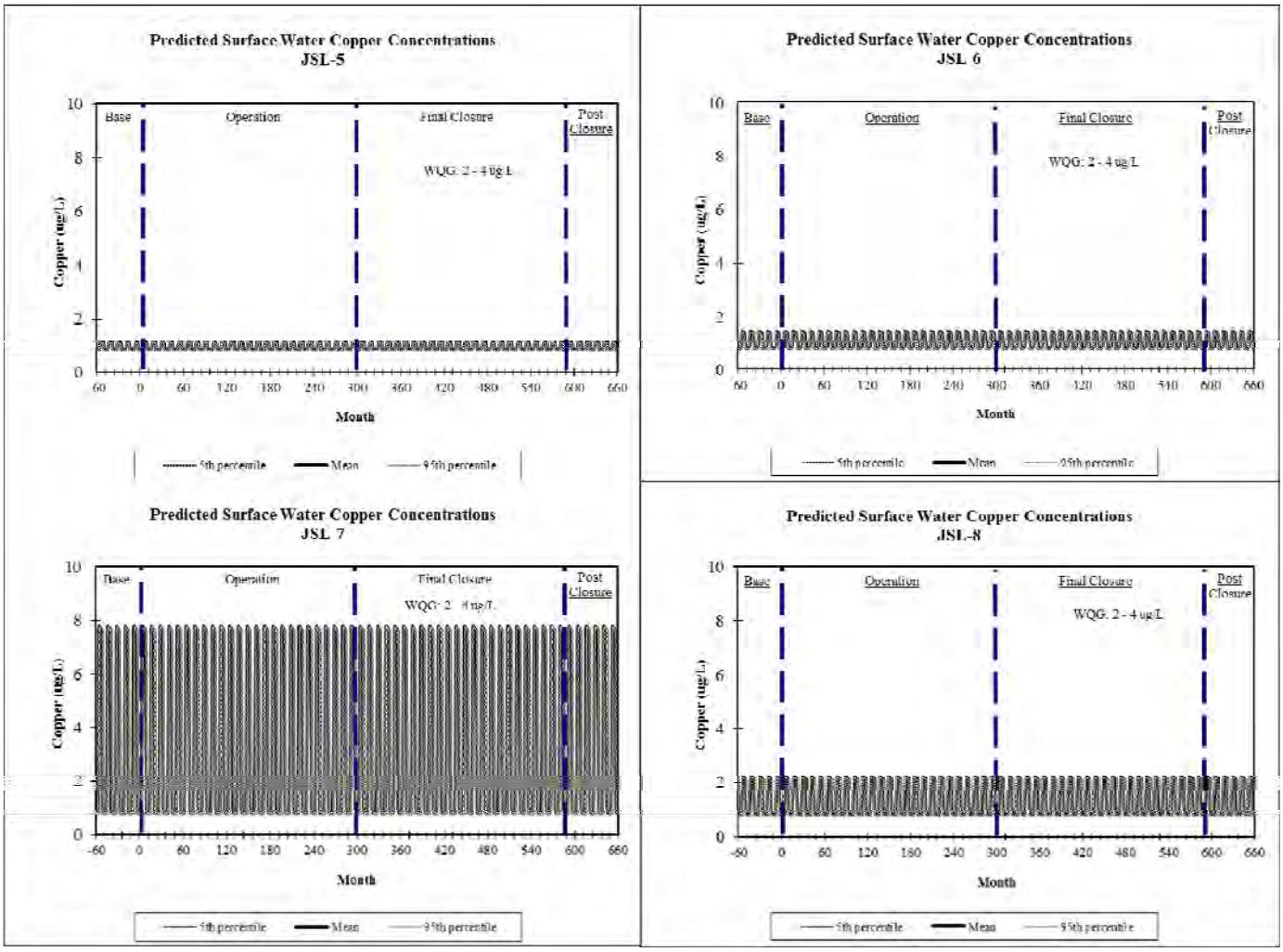


Figure 7.2-1 Water Quality Predictions (Cont'd)

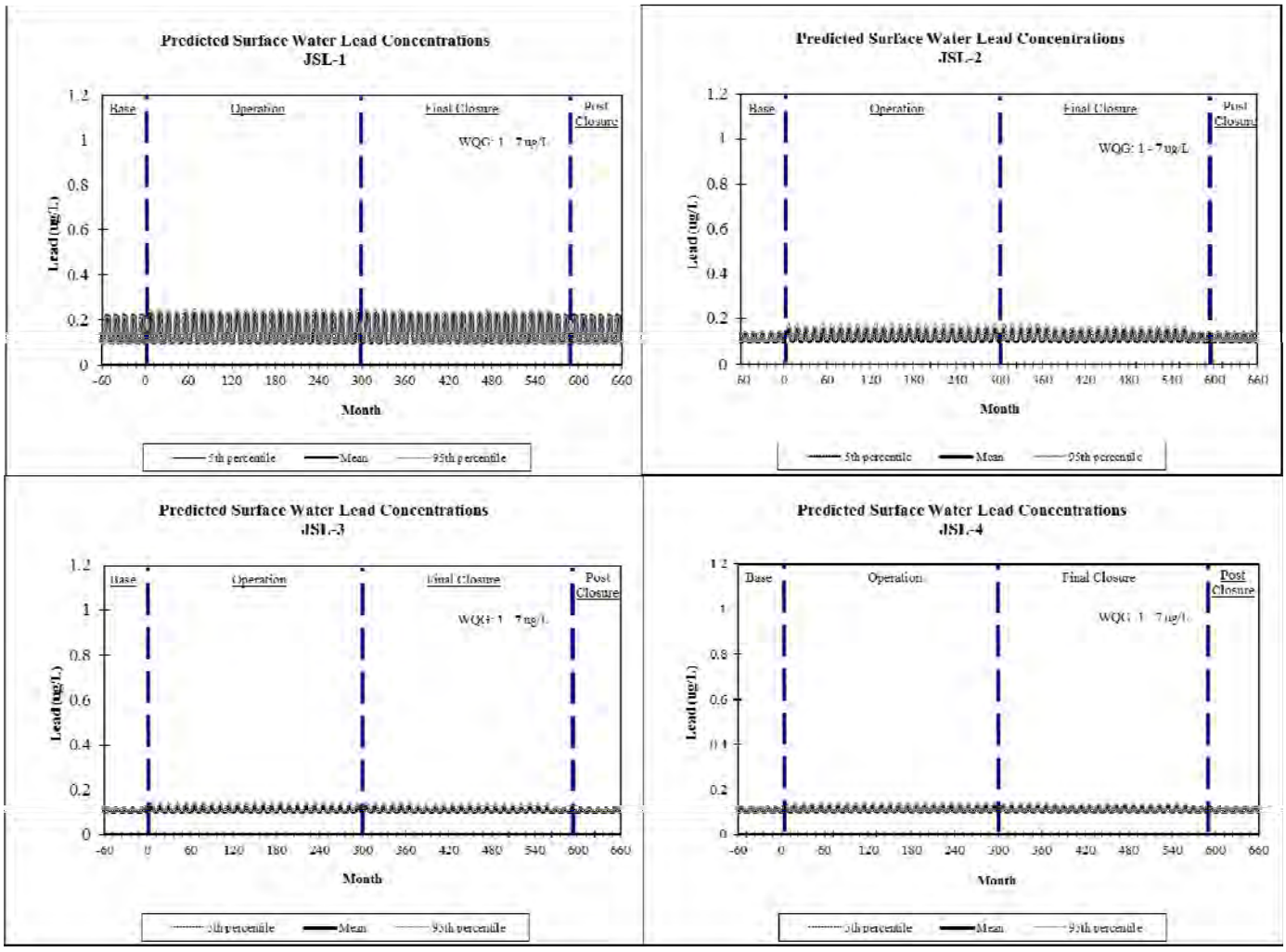


Figure 7.2-1 Water Quality Predictions (Cont'd)

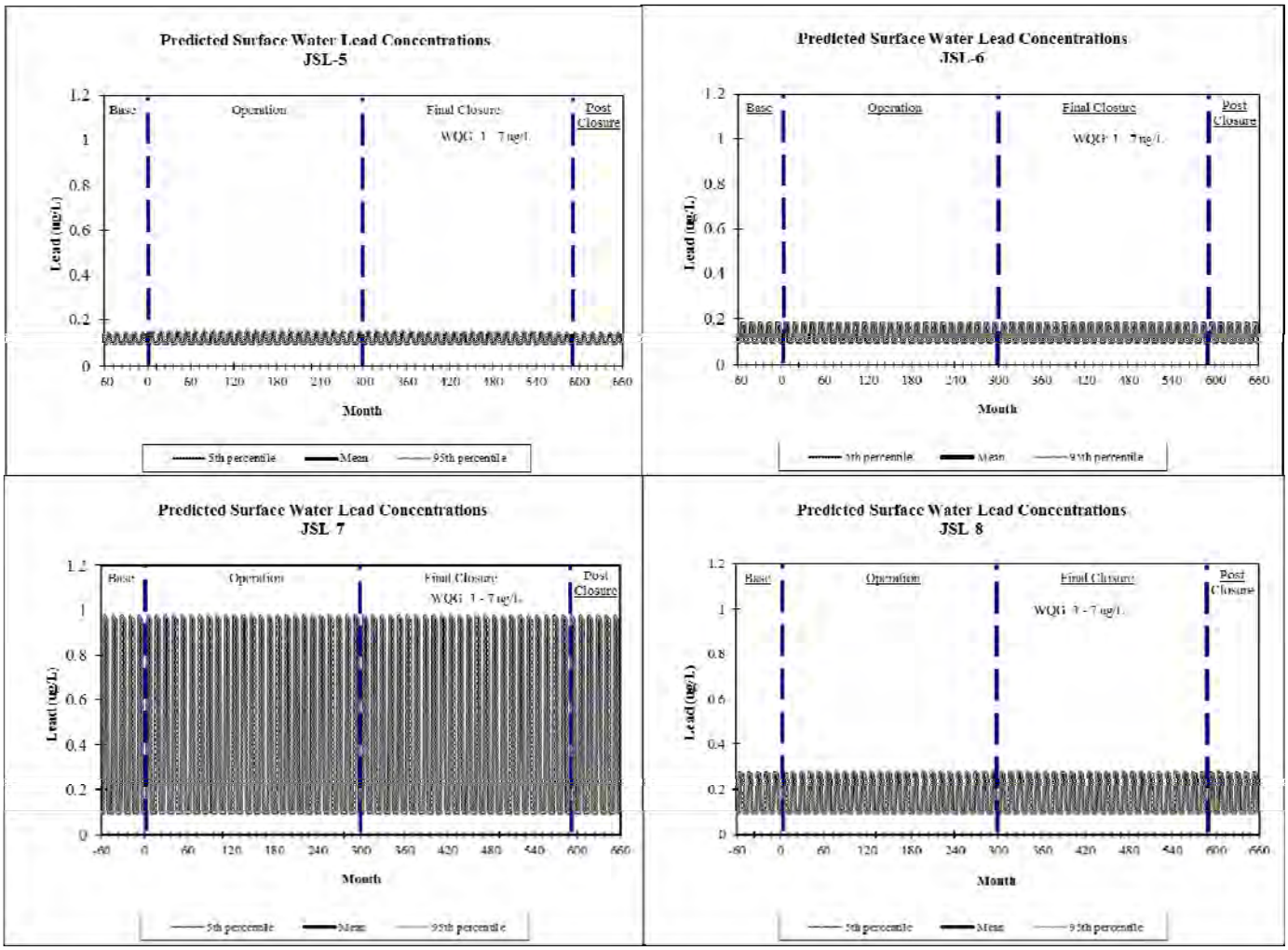


Figure 7.2-1 Water Quality Predictions (Cont'd)

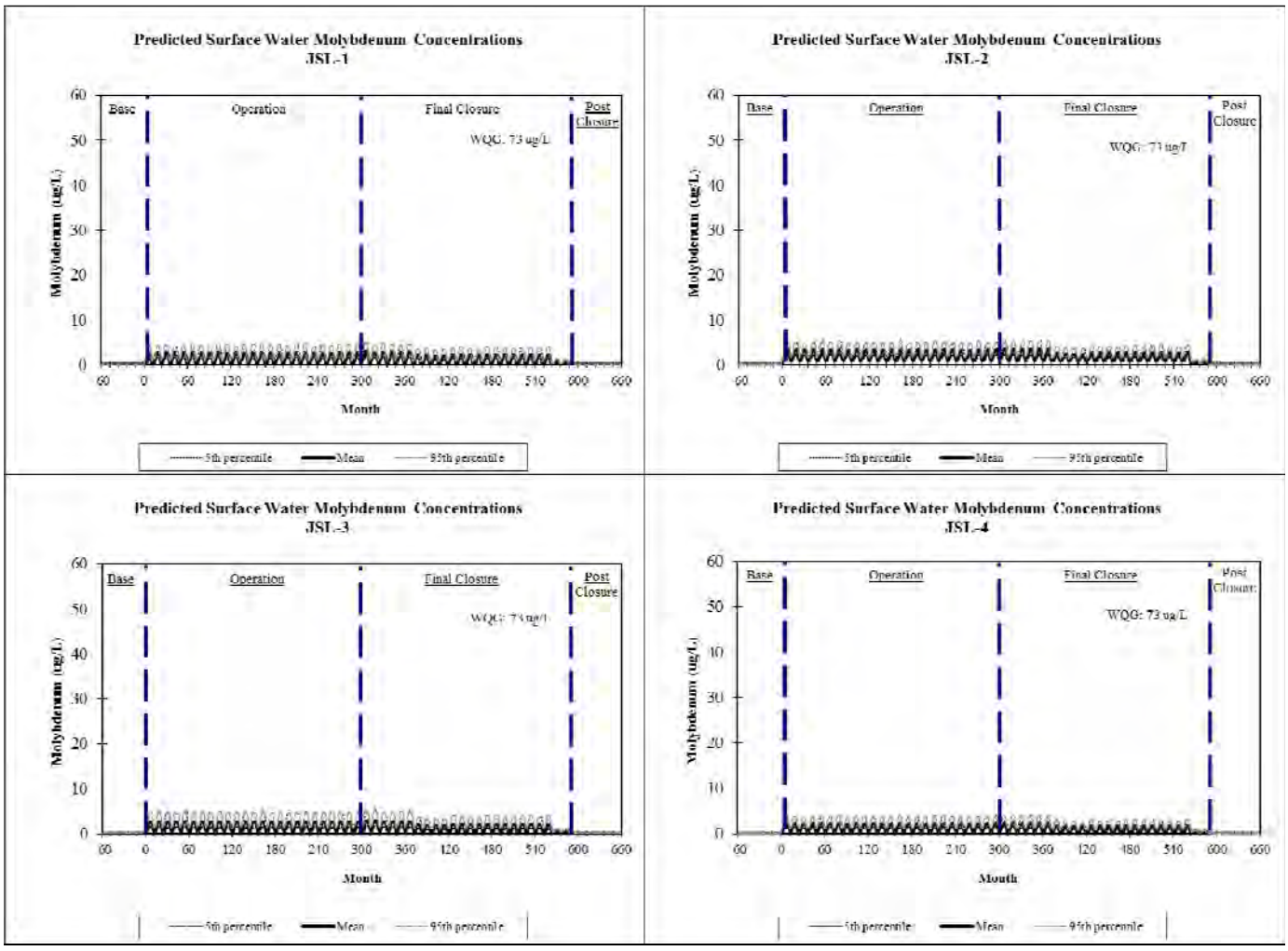


Figure 7.2-1 Water Quality Predictions (Cont'd)

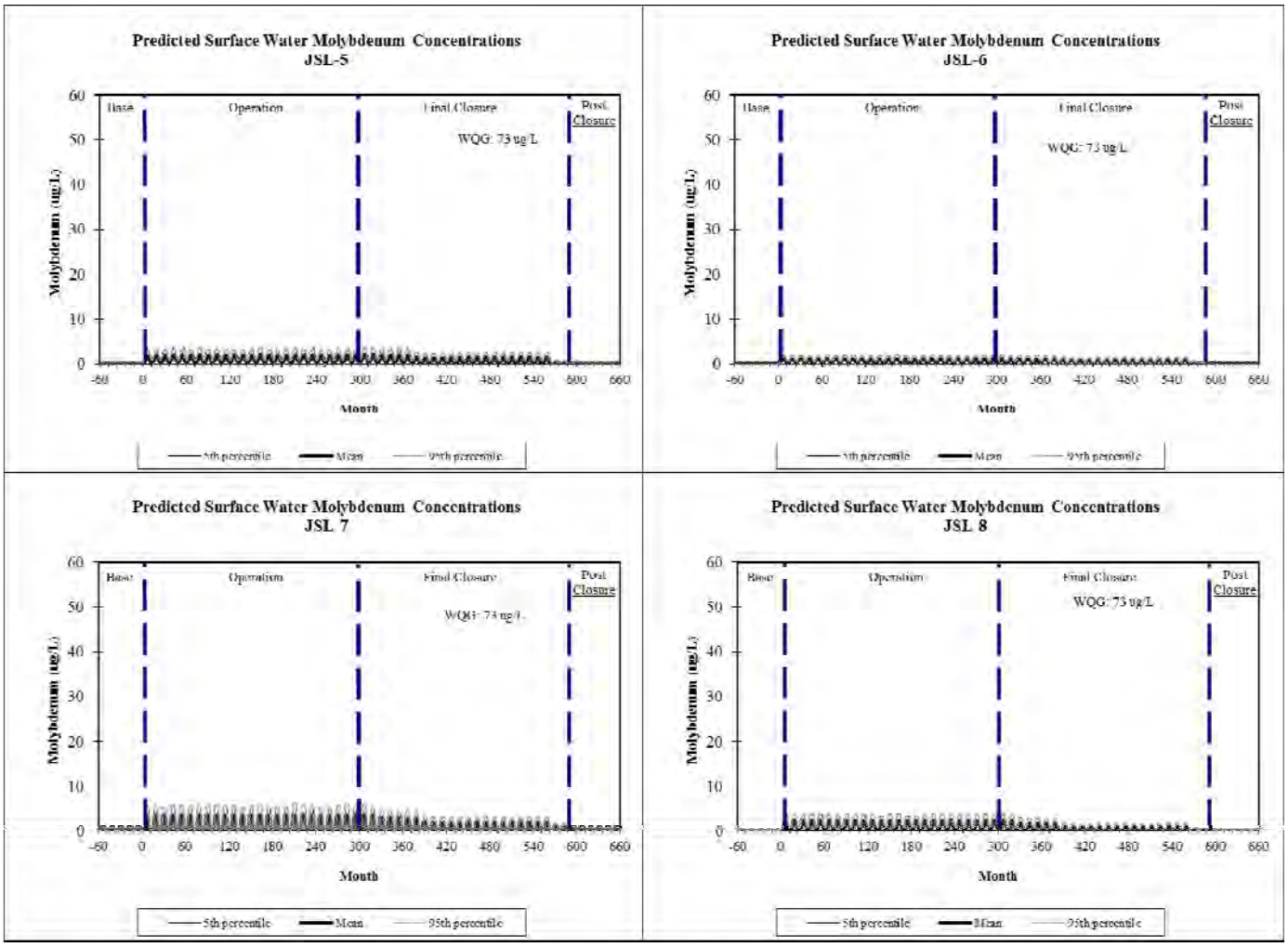


Figure 7.2-1 Water Quality Predictions (Cont'd)

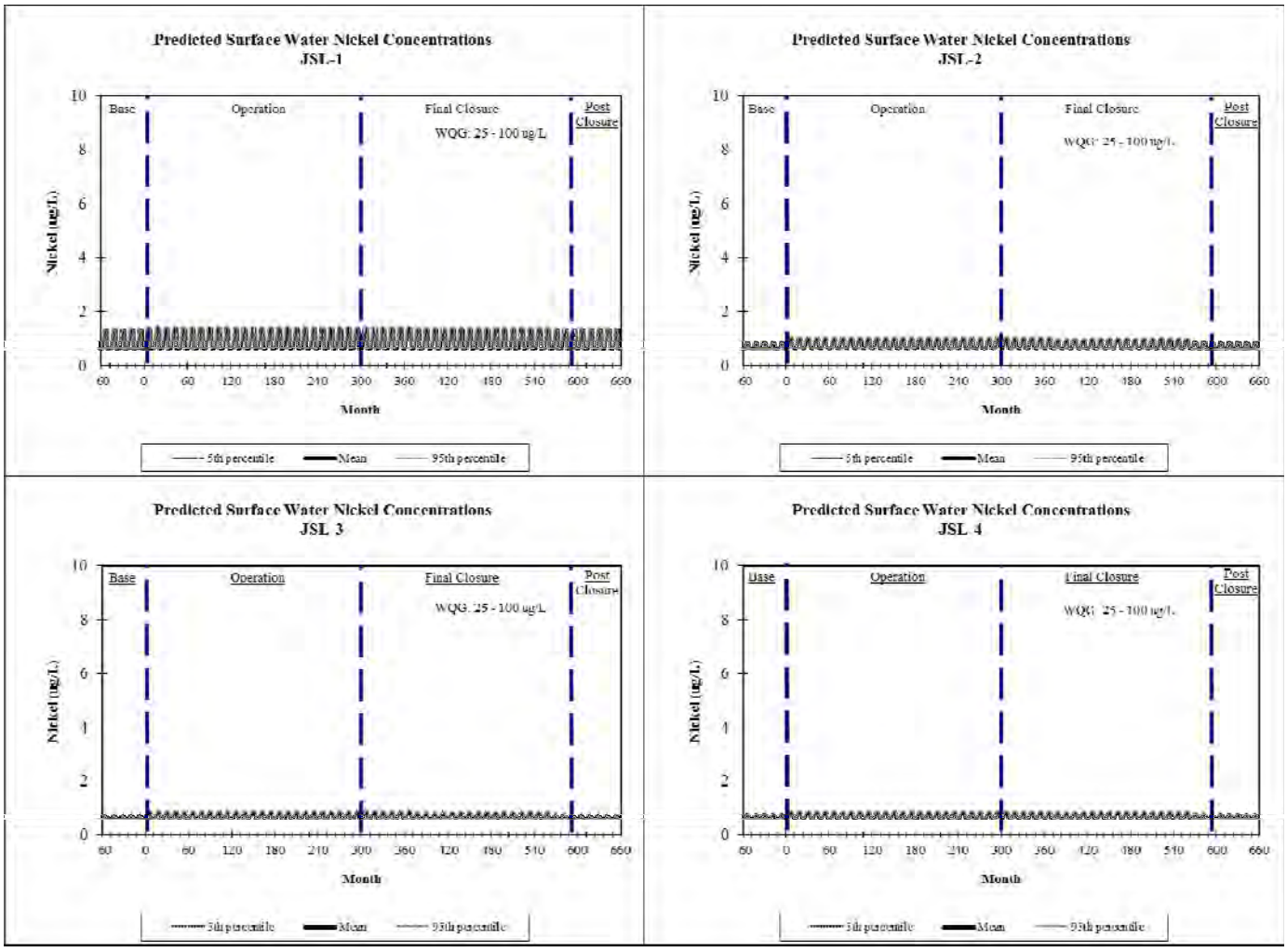


Figure 7.2-1 Water Quality Predictions (Cont'd)

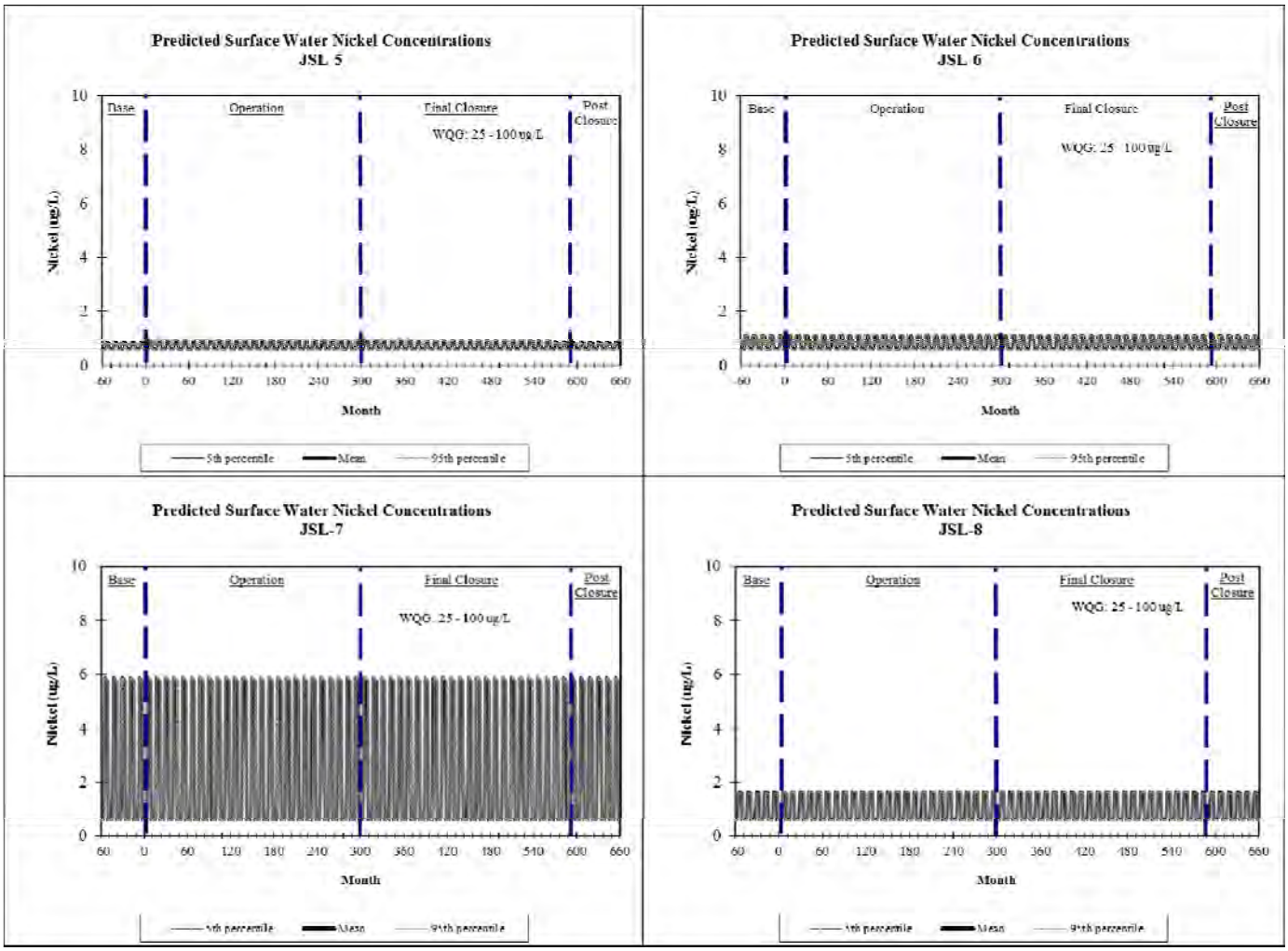


Figure 7.2-1 Water Quality Predictions (Cont'd)

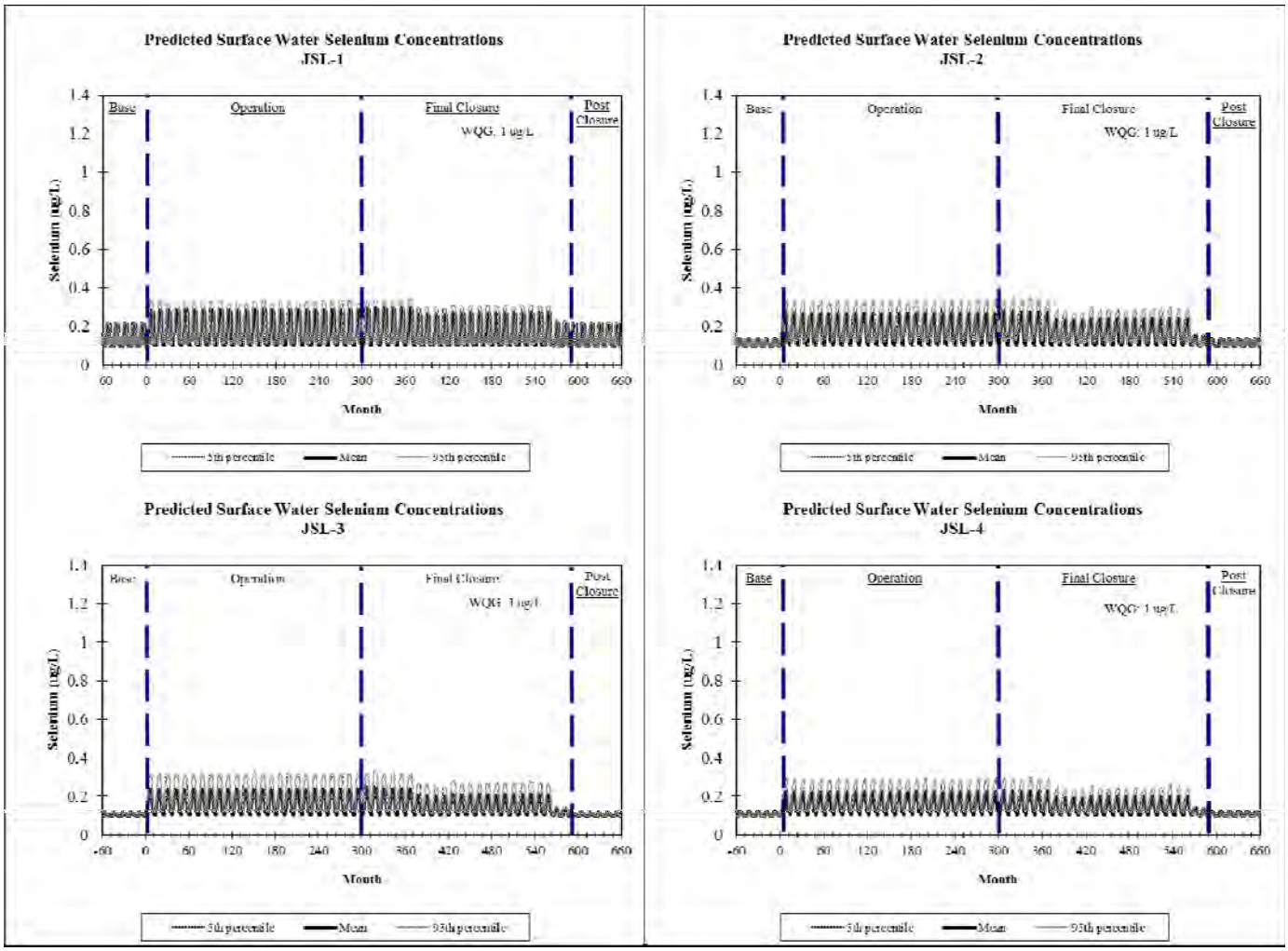


Figure 7.2-1 Water Quality Predictions (Cont'd)

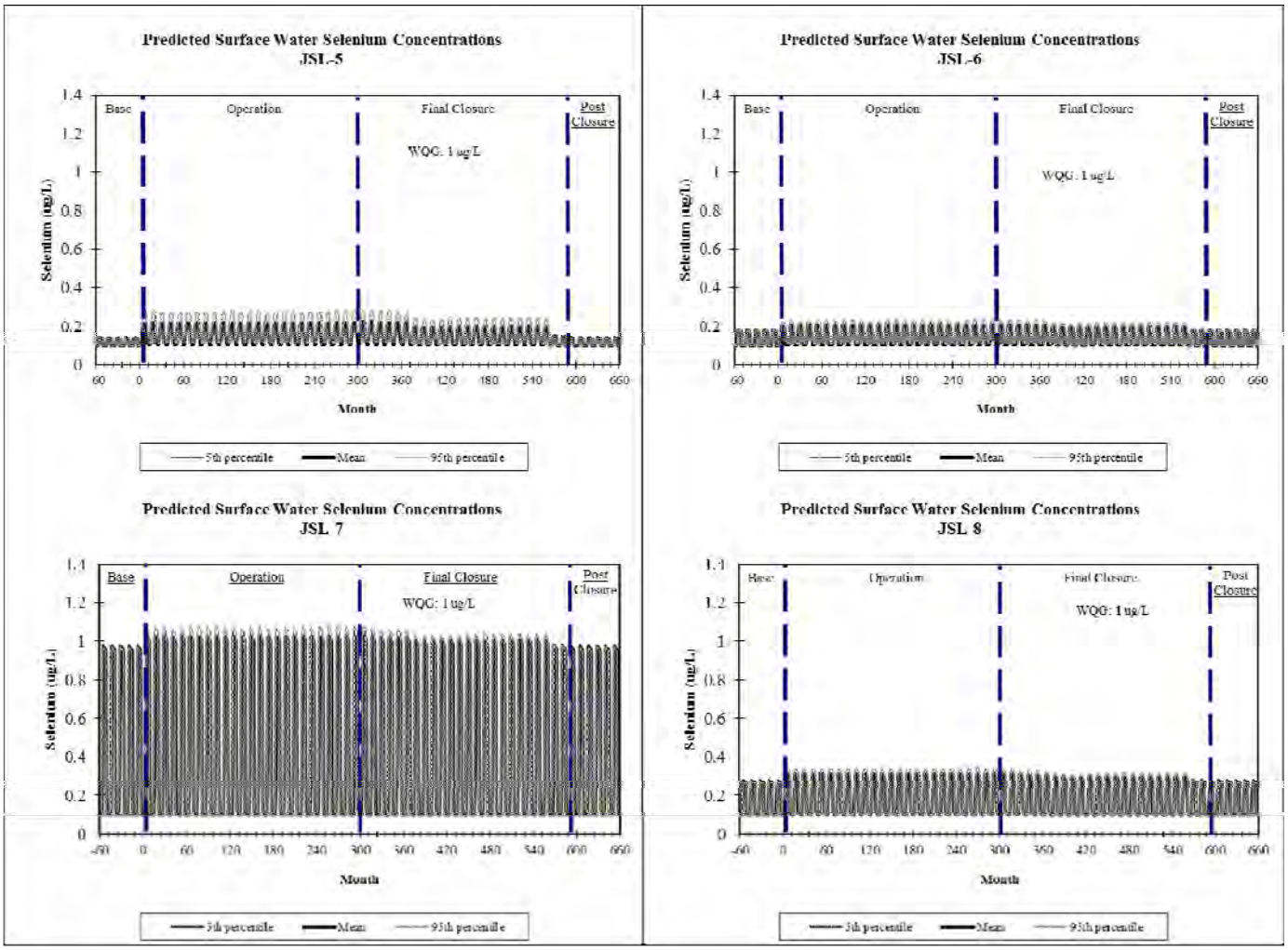


Figure 7.2-1 Water Quality Predictions (Cont'd)

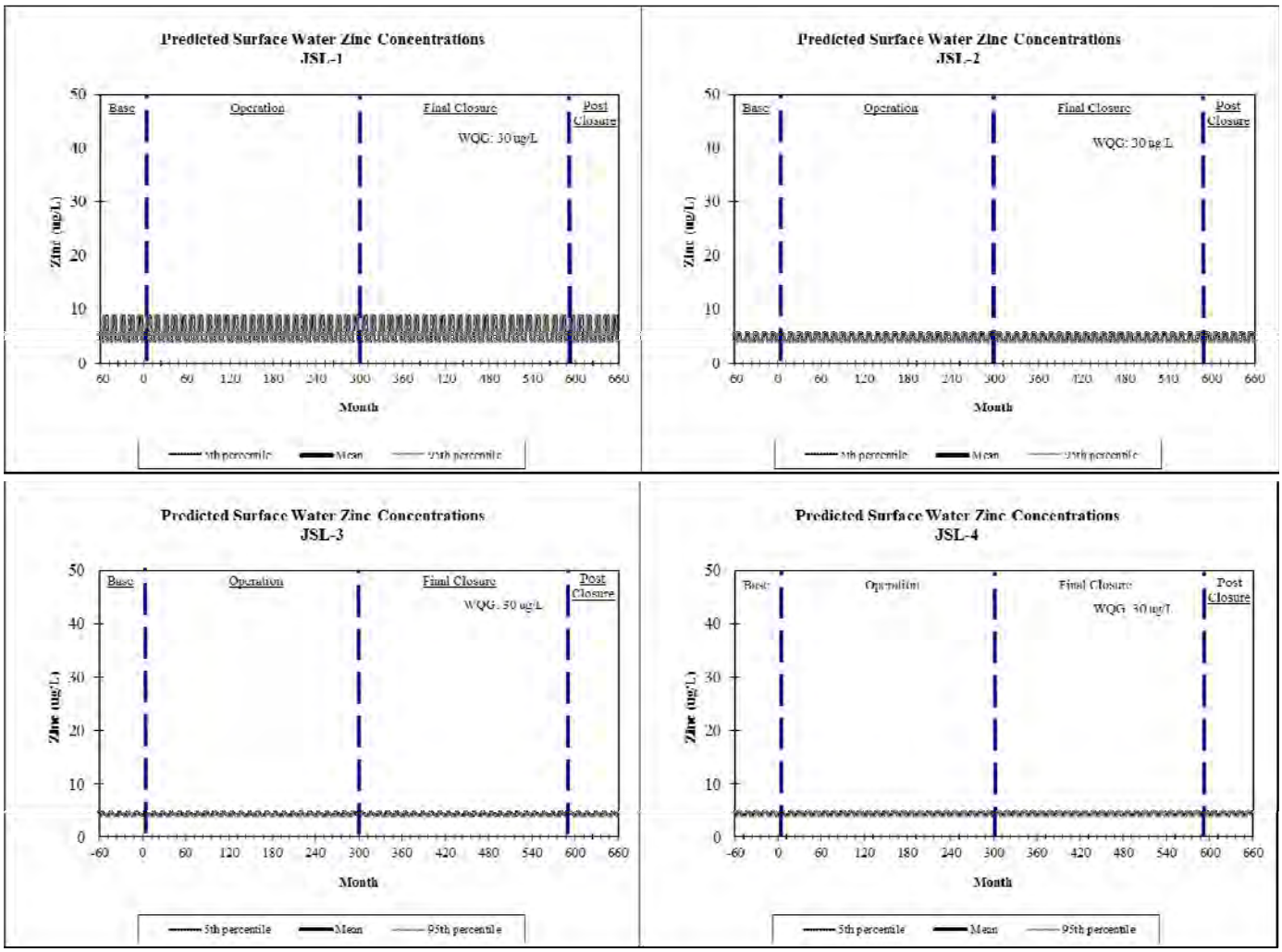


Figure 7.2-1 Water Quality Predictions (Cont'd)

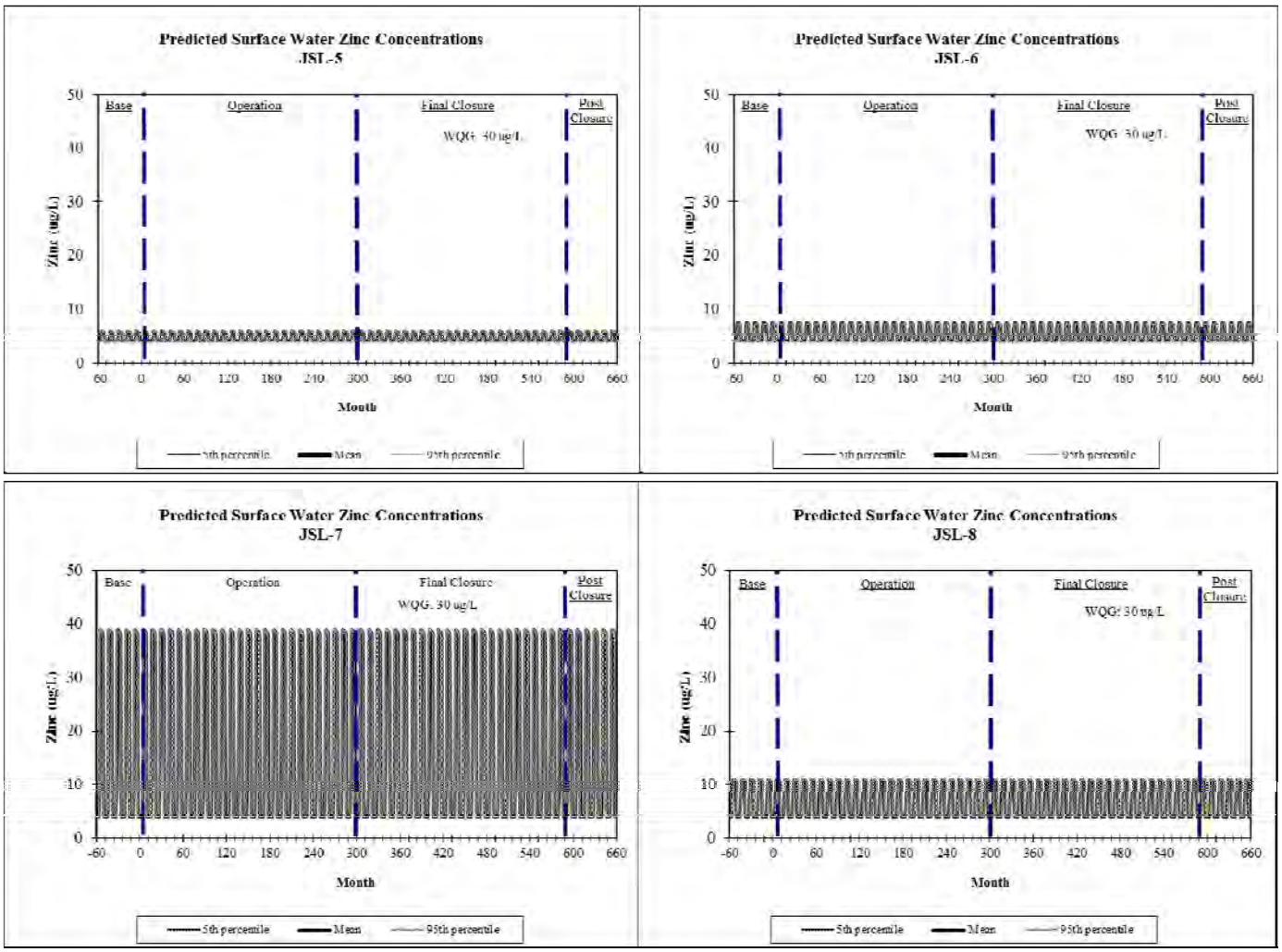


Figure 7.2-1 Water Quality Predictions (Cont'd)

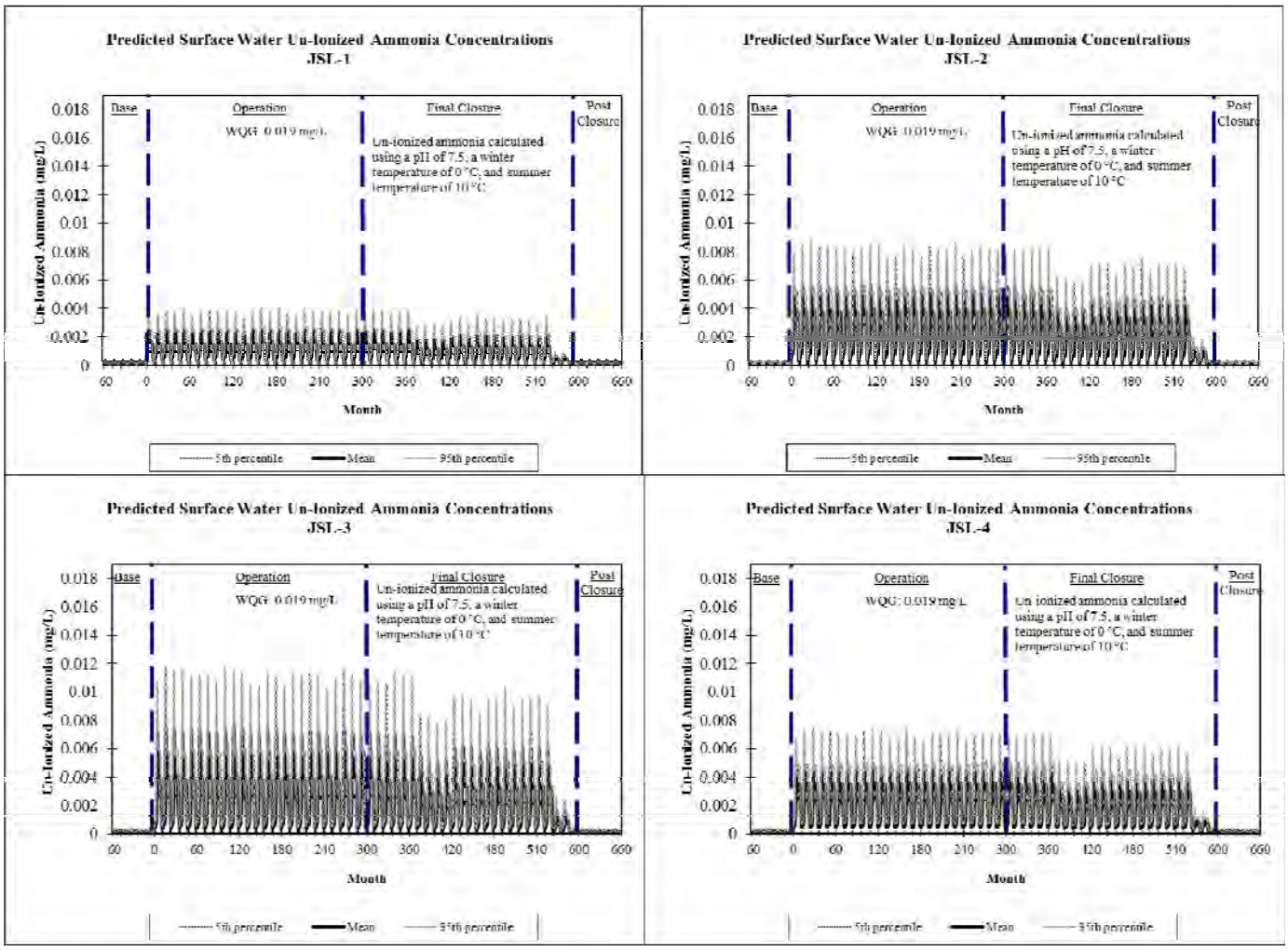


Figure 7.2-1 Water Quality Predictions (Cont'd)

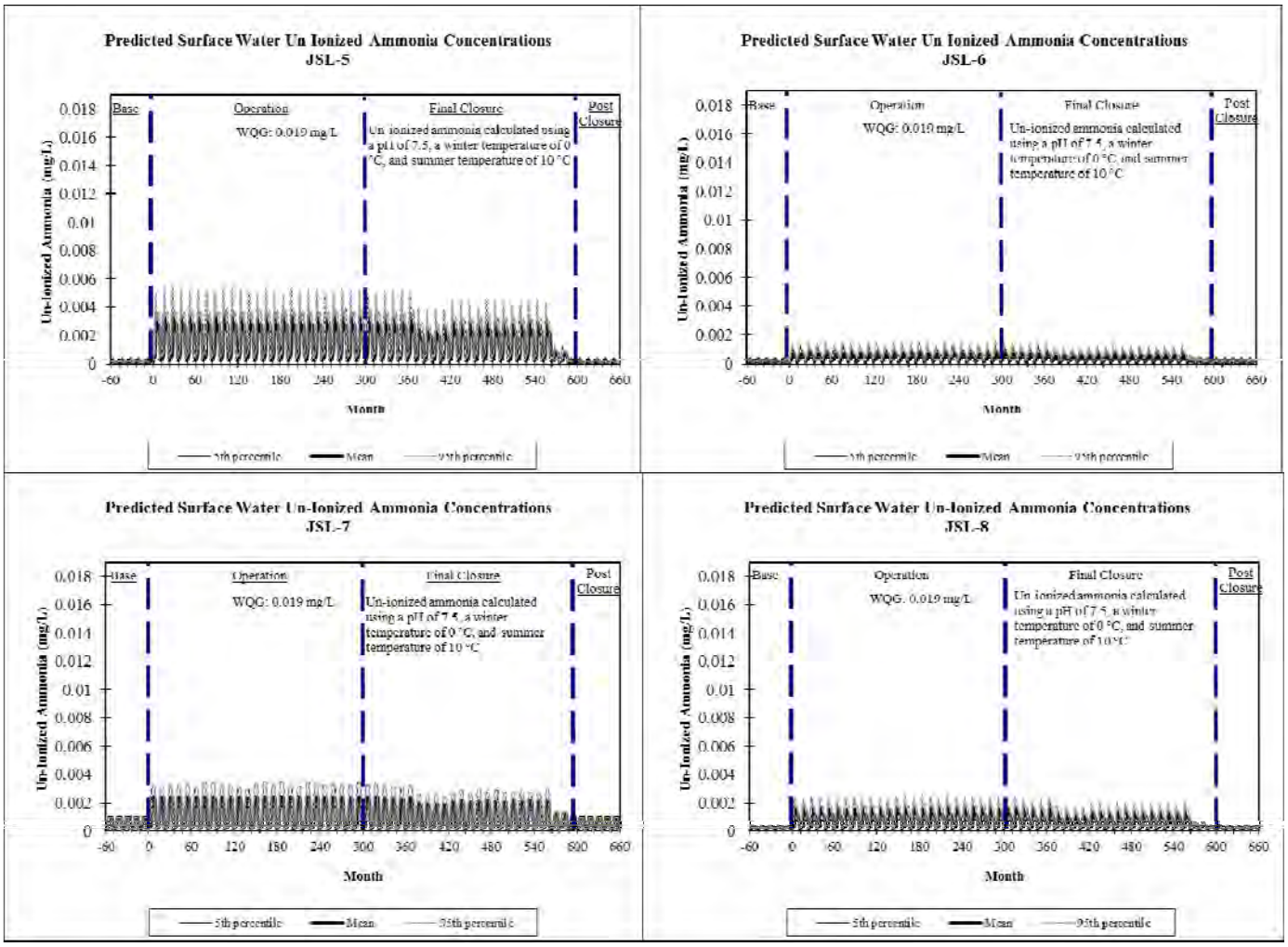


Figure 7.2-1 Water Quality Predictions (Cont'd)

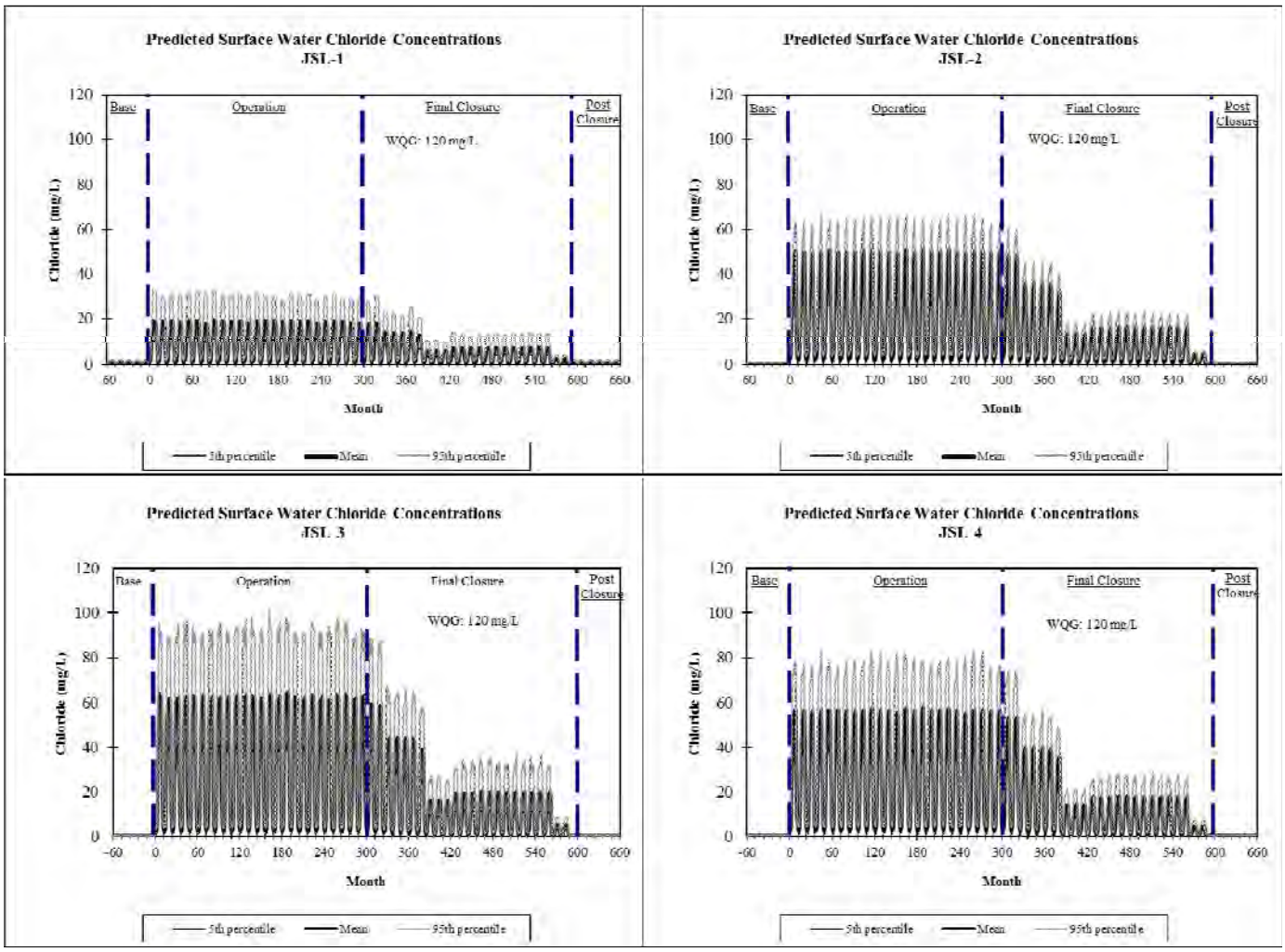


Figure 7.2-1 Water Quality Predictions (Cont'd)

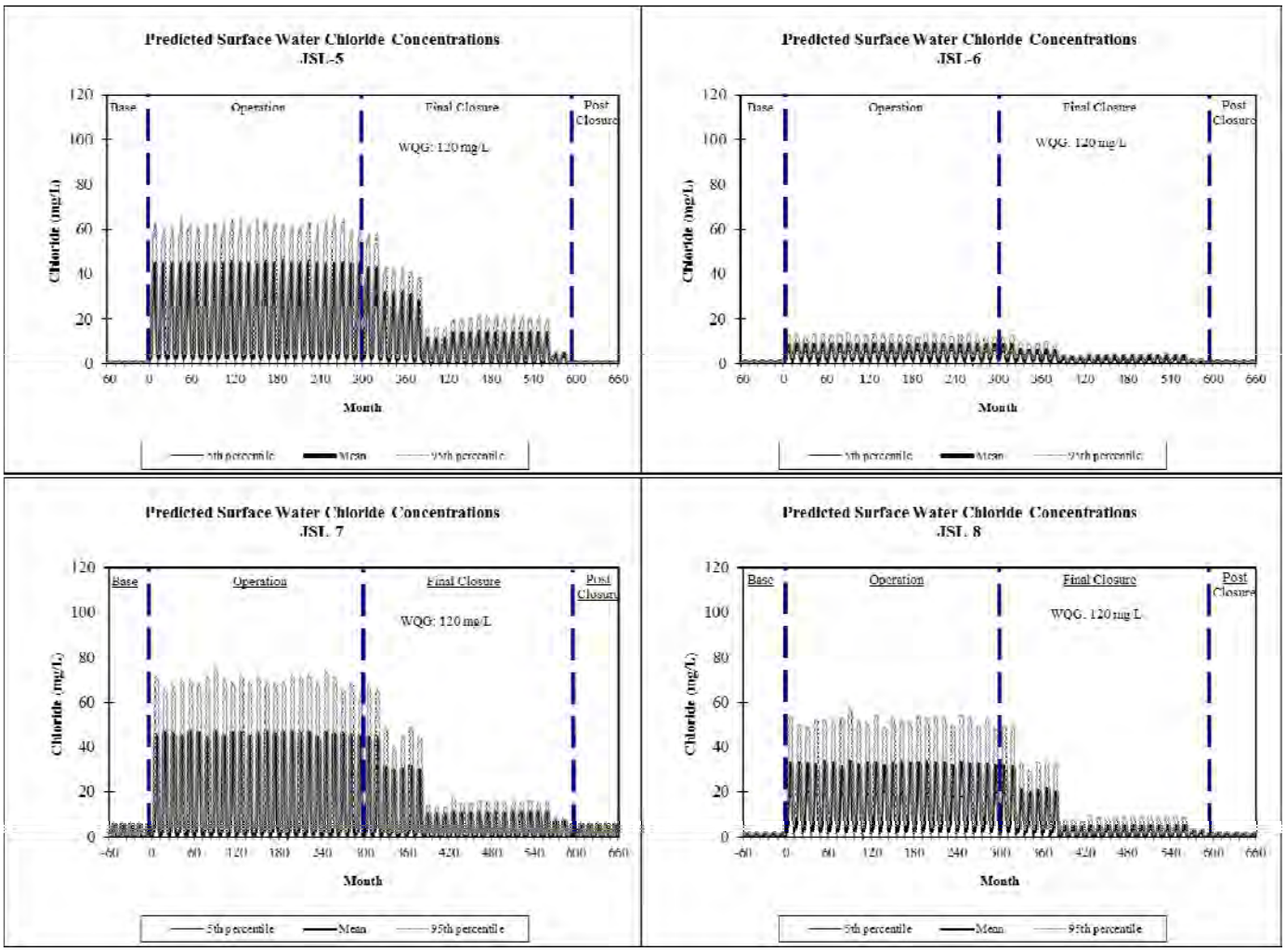


Figure 7.2-1 Water Quality Predictions (Cont'd)

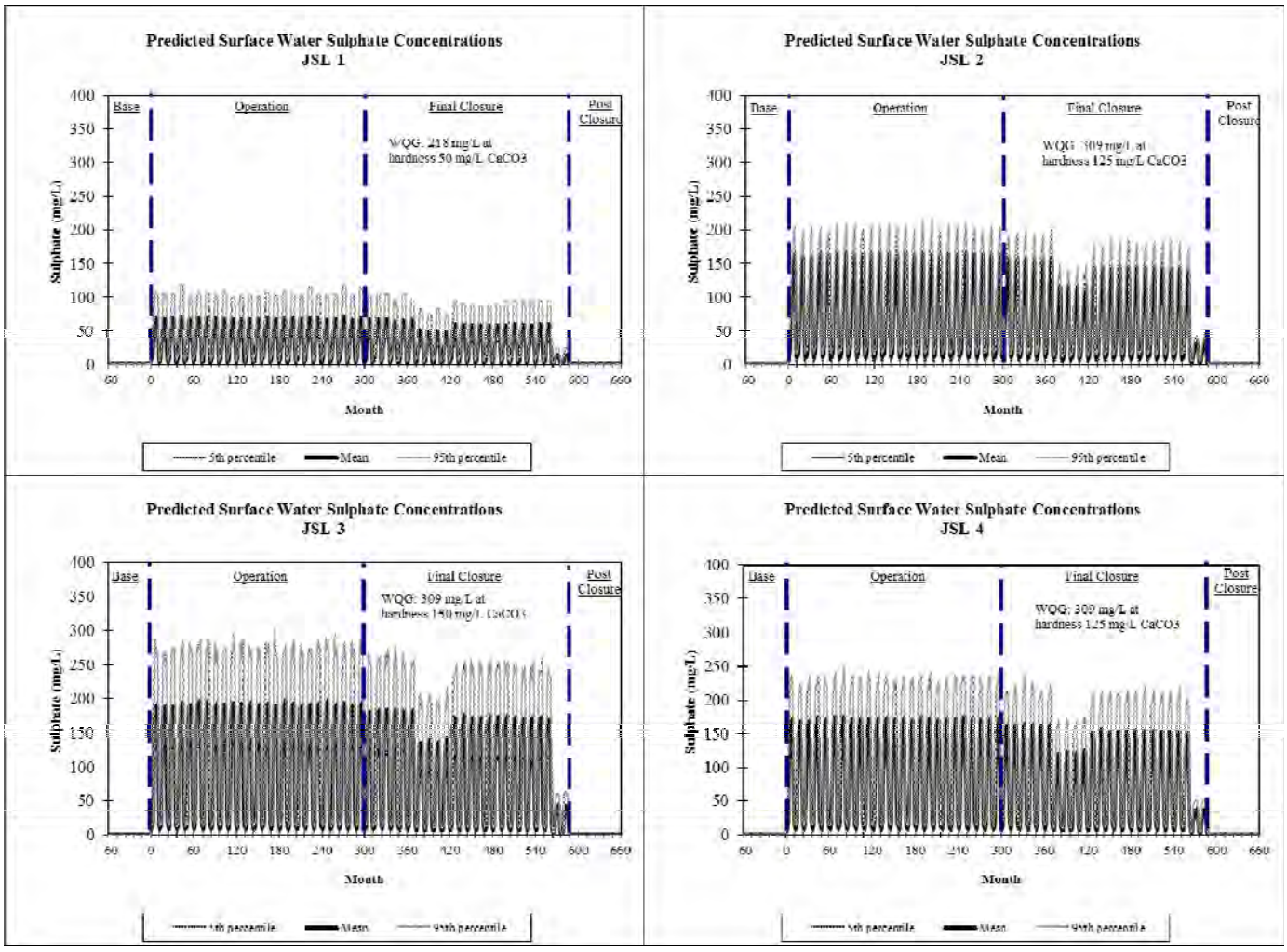


Figure 7.2-1 Water Quality Predictions (Cont'd)

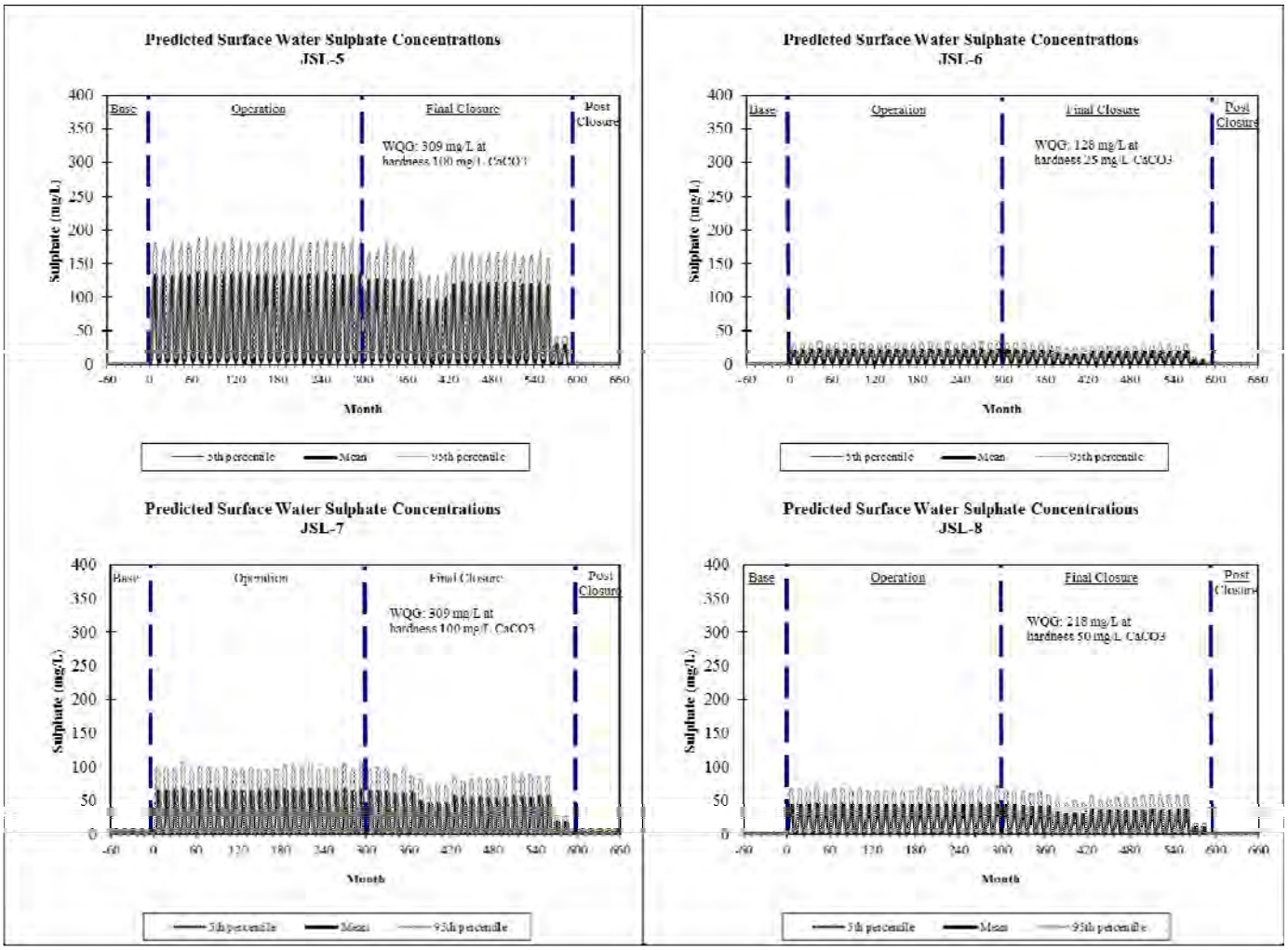


Figure 7.2-1 Water Quality Predictions (Cont'd)

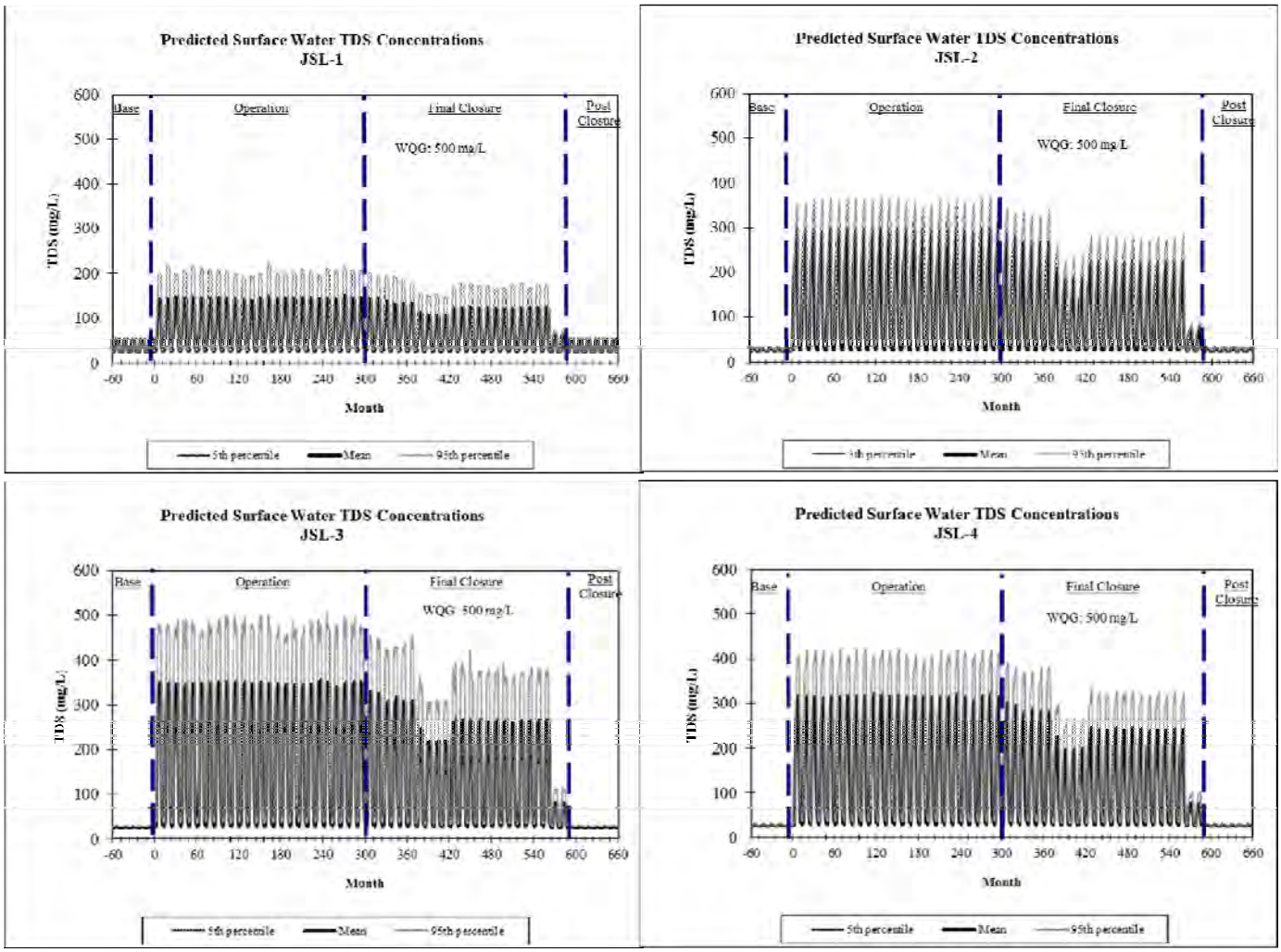


Figure 7.2-1 Water Quality Predictions (Cont'd)

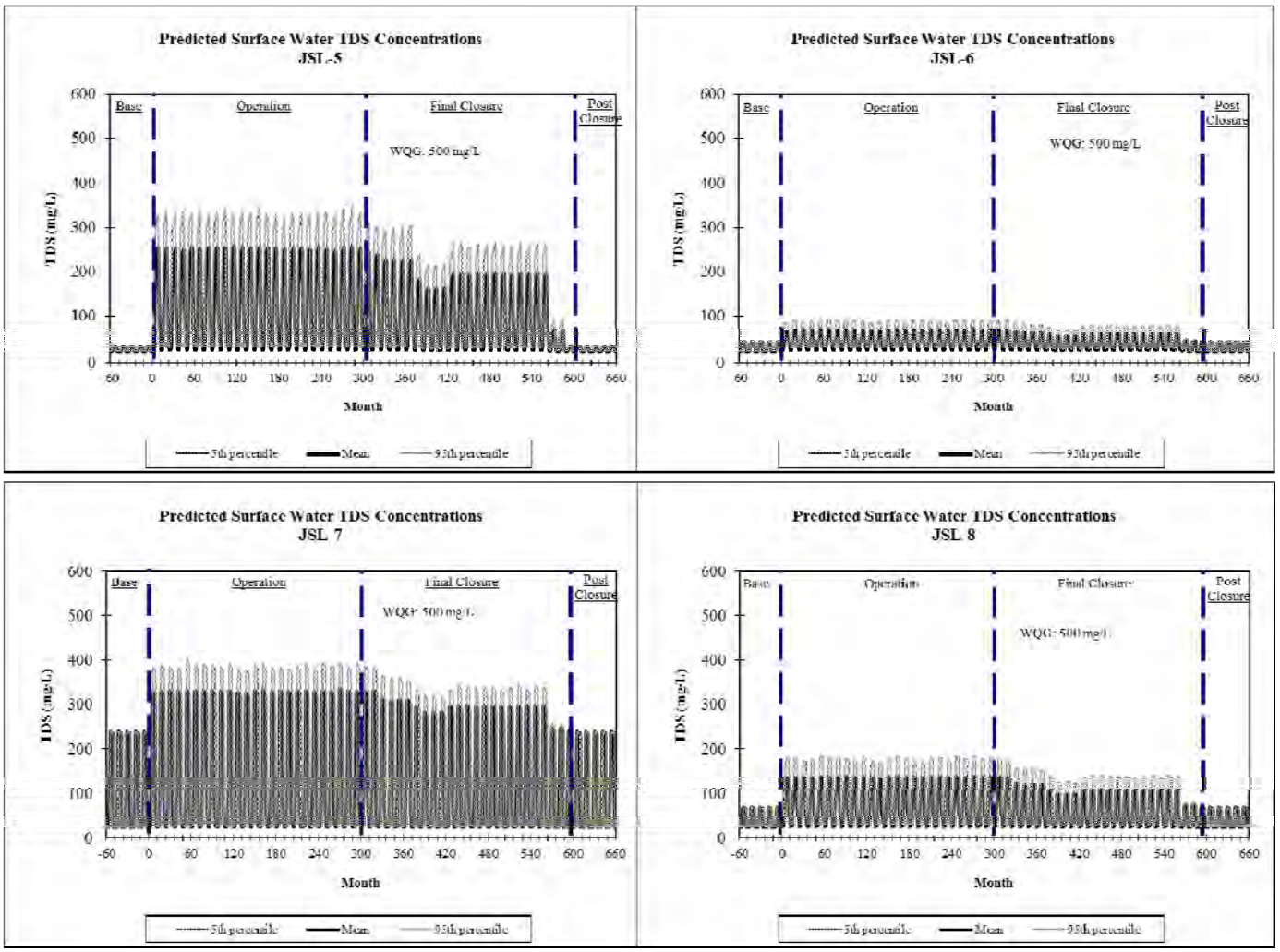


Figure 7.2-1 Water Quality Predictions (Cont'd)