9.5 AQUATIC ORGANISMS AND HABITAT

9.5.1 Baseline

9.5.1.1 Introduction

Aquatic baseline information was collected from Snap Lake, a reference lake and several small lakes within or near the project footprint

An aquatic baseline study was initiated in 1998 and continued in 1999 and 2001 to characterize baseline environmental conditions for the Snap Lake Diamond Project. The studies documented existing aquatic information on pre-development conditions of surface waterbodies potentially affected by the Snap Lake Diamond Project. Waterbodies included Snap Lake as well as small lakes and streams near the Snap Lake Diamond Project. Aquatic information was also collected for an initial reference lake in 1998 (the north lake) considered comparable at the time to Snap Lake. A second reference lake was selected in 1999 after the physical features of the first reference lake proved to be different from Snap Lake. MacKay Lake fish were sampled in 2001 to provide data from an additional regional fish population.

Zooplankton, phytoplankton, and benthic invertebrate samples were collected from a variety of locations

Zooplankton and phytoplankton sampling was conducted in Snap Lake and the reference lake in summer and fall of 1999 (Figure 9.5-1). Benthic invertebrate sampling was carried out in September 1999 (Figure 9.5-2). Sediment samples were collected in September 1999 to determine baseline metal levels in Snap Lake and the reference lake. Appendix IX.5 and IX.6 contain detailed methods and results for sediment sampling.

Fish and fish habitat information was also collected from a number of lakes

Fish habitat mapping was completed in 1999 and 2001 for nine inland lakes (identified by the prefix IL), four north inland lakes (identified by the prefix NL), and Snap Lake. Fish population information was also collected from these lakes, as well as the reference lake and MacKay Lake. Fish health information was limited to sampling from Snap Lake, the reference lake, and MacKay Lake.

9.5.1.2 Methods

9.5.1.2.1 Introduction

Detailed methods used to collect baseline data can be found in Appendices IX.5 and IX.9

The following summary outlines the methods used for the collection of samples relevant to the assessment of baseline conditions for aquatic organisms and habitat related to the Snap Lake Diamond Project. For a full description of the sediment and aquatic organisms sampling methods, refer to Appendices IX.5 and IX.9.

Figure 9.5-1 Zooplankton and Phytoplankton Sampling Locations for Snap Lake and the Reference Lake (1999)

Figure 9.5-2 Benthic Invertebrate Sampling Locations for Snap Lake and the Reference Lake (1999)

9.5.1.2.2 Phytoplankton and Zooplankton Sampling

Phytoplankton community and chlorophyll* a *samples were collected as well as zooplankton community and biomass samples

Samples for the phytoplankton community and chlorophyll *a* (chl *a*) analysis were collected from Snap Lake and the reference lake in 1999. Concurrently, zooplankton samples were collected at the same locations and at a number of shallow water sites in areas potentially used by small fish (Figure 9.5-1). Zooplankton sampling was undertaken to assess biomass and community structure. Phytoplankton and zooplankton collection methods for nutrients and community structure are outlined in Golder Associates Ltd. Technical Procedure 8.7-1 (Appendix IX.9).

9.5.1.2.3 Benthic Invertebrate Community Sampling

Benthic invertebrate samples were collected in Snap Lake and the reference lake

The benthic invertebrate community sampling program was carried out in Snap Lake at three sites around the northwest peninsula and one site in the eastern portion of the lake in 1999 (Figure 9.5-2). Four sites were sampled in the reference lake. All sites were close to the shore (*i.e.*, within 500 metres [m]) and were collected at a similar depth (6 m to 7 m). Qualitative kicknet samples were also collected along the shoreline in the vicinity of each benthos sampling site if shoreline vegetation was present. This was done to ensure that all common invertebrates species were captured. In addition, field information such as water depth was collected at each location. Benthic invertebrate collection methods are described in detail in Golder Associates Ltd. Technical Procedures 8.2-2 (Appendix IX.9).

Benthic invertebrates were identified using published keys; quality control measures were undertaken

Benthic invertebrates were identified to the lowest practical taxonomic level, which was usually genus. Damaged or very small specimens were typically identified to the family level. Identifications were made using published keys and with comparisons to reference material. A quality control program was established to ensure consistency of results during sample analysis (Appendix IX.9).

Benthic invertebrate data are expressed as total abundance and taxonomic richness

The total abundance, taxonomic richness, and mean percentage of total abundance of each major invertebrate group at each site were calculated. Richness was calculated as the total number of taxa present at each site, and as the number of taxa collected in the qualitative shoreline kicknet samples. Supporting data directly relevant to benthic samples (water depth, sediment particle size, total organic carbon [TOC], and total inorganic carbon [TIC]) were also tabulated for each sampling site.

9.5.1.2.4 Fisheries Sampling

The objective of the fisheries baseline study, conducted in 1998, 1999 and 2001, was to define environmental baseline conditions in the Snap Lake area

The objective of the fisheries baseline study was to characterize baseline environmental conditions of the Snap Lake area that may be affected by the Snap Lake Diamond Project. Initiated in 1998, the program was expanded in 1999. In 2001, additional baseline information was collected in relation to a small number of specific project components. Waterbodies in the survey included Snap Lake, small lakes and streams near the project, the reference lake, and MacKay Lake (Figures 9.5-3 and 9.5-4). Table 9.5-1 summarizes the components of the fisheries survey.

Table 9.5-1 Components of the 1999 and 2001 Fisheries Survey

Waterbody	Survey Details
Snap Lake	• shoreline lake habitat mapping • bathymetry survey • collection of muscle and liver tissue for metals analysis from two species of fish (round whitefish and lake trout) • general non-lethal fish inventory, building on 1998 inventory data • sampling for fish in areas of potential rearing habitat • fall lake trout spawning survey and habitat assessment
Reference Lake	• collection of fish muscle and liver tissue for metals analysis from two species of fish: round whitefish (*Prosopium cylindraceum)* and lake trout (*Salvelinus namaycush*) • sampling for fish in areas of potential rearing habitat
MacKay Lake	• collection of fish muscle and liver tissue for metals analysis from two species of fish (round whitefish and lake trout)
IL (2 to 9) **NL** (1 to 4)	• lake bathymetry and shoreline habitat mapping • general non-lethal fish inventory
Snap Lake Streams (inlet and outlet tributaries to and from Snap Lake)	• fish habitat survey at peak and low flows • fish inventory or observations of fish in (or near) streams • kick sampling for eggs and stream habitat use

IL1 was surveyed in the summer of 1999 and de-watered in March 2000

IL1 was surveyed in 1998 and 1999. In late 1999, IL1 was approved as the location of the processed kimberlite containment (PKC) area for the Advanced Exploration Program (AEP) at Snap Lake. The PKC has since been renamed the water management pond (WMP) for this application. The lake was de-watered in March 2000. Results of fish habitat and bathymetry for this waterbody and the associated outlet stream (S30) were presented in an earlier report to the Northwest Territories (NWT) Water Board and to Fisheries and Oceans Canada (DFO) and will not be repeated in this report.

Figure 9.5-3 Streams and Inland Lake Fisheries Sampling Locations (1999, 2001)

Figure 9.5-4 Reference and Regional Lake Fisheries Sampling Locations (1999, 2001)

Fish Inventory

Fish were captured by gillnets, angling, minnow traps, seine nets, and electrofishing; nets and traps were set in a variety of habitats

In 1999, a fish inventory was conducted on Snap Lake, the reference lake, IL 2 to 6, and NL 1 to 4. IL6 was re-sampled in 2001. IL 7 to 9 were inventoried in July 2001. Fish were captured by gillnets, angling (spin-cast and fly-fishing), minnow traps, seine nets, and electrofishing. Gill nets and minnow traps were set in a variety of different habitat types in each lake. Fishing was conducted according to Golder Associates Ltd. Technical Procedure 8.1-3. A number of parameters such as fish length and weight were recorded from each captured fish (Appendix IX.9).

Fish Tissue Collection

Muscle and liver tissue were collected from lake trout and round whitefish for determination of metal concentrations

Adult fish from two levels of the aquatic food chain were collected from Snap Lake, the reference lake, and MacKay Lake. Lake trout *(Salvelinus namaycush)* were collected to represent top predators, and round whitefish *(Prosopium cylindraceum)* were collected to represent secondary consumers. A maximum of 10 lake trout and 15 round whitefish were collected from each of the three lakes. Muscle and liver tissue were collected from each fish and were analyzed for total metal concentrations. Tissue samples were removed from each fish according to Golder Associates Ltd. Technical Procedure 8.16 (Appendix IX.9).

Lake Trout Spawning Survey

Fall fish spawning surveys were conducted in Snap Lake to identify the location of spawning habitat for lake trout; the majority of lake trout captured were returned to Snap Lake

Fall fish spawning surveys were conducted to identify the location of spawning habitat of lake trout in Snap Lake. Lake trout were collected to establish the location and usage of habitat by spawning fish. Suspected spawning sites were visually inspected for congregations of adult lake trout. Angling and limited gill netting were used to assess the density of spawning fish. The catch per unit effort (CPUE) was calculated as the number of fish caught per hour per person of angling effort. The majority of the captured fish were returned to Snap Lake after determination of weight, length, sex (if possible), spawning condition, and collection of a non-lethal aging structure (Appendix IX.9).

9.5.1.2.5 Bathymetry

Snap Lake, IL 2 to 9, and NL 1 to 4 were surveyed to prepare bathymetric maps

Snap Lake, IL 2 to 6, and NL 1 to 4 were surveyed in 1999. IL 7 to 9 were surveyed in July 2001. The bathymetry of each lake was recorded along transects with a chart-recording echo-sounder. Data from the echo-sounder were used to create a bathymetric map of each lake.

9.5.1.2.6 Fish Habitat Mapping

Habitat mapping was conducted for Snap Lake, IL 2 to 4 and 6 to 9, and NL 1 to 6

Habitat mapping of Snap Lake, IL 2 to 4, and NL 1 to 6 was conducted in 1999. IL 7, 8, and 9 were surveyed in July 2001. Habitat mapping was conducted according to Golder Associates Ltd. Technical Procedures 8.19-0 (Appendix IX.9). Habitat mapping was done by visual estimation. Shoreline characteristics were recorded on an enlarged 1:50,000 scale map of each lake. Photographs of each lake were taken. Appendix IX.9 lists the specific characteristics of the shoreline (terrestrial) and nearshore (shoreline to a depth of 4 m) habitat areas that were mapped.

9.5.1.2.7 Stream Surveys

Streams were surveyed to gauge relative stream habitat availability

The intent of the stream habitat surveys was to collect information on habitat capability, stream accessibility, fish species presence, and fish use of the streams. Streams within the Snap Lake watershed were surveyed to gauge the relative stream habitat availability for the fish community (Figure 9.5-3).

Streams in the Snap Lake watershed were assessed for fish habitat potential

In May 1999, based on 1:50,000 topographic maps and a subsequent aerial survey, 30 inlet streams and two outlet streams were identified in the Snap Lake watershed. Some of the areas denoted as streams on the topographic map were small, ponded areas with no flow or channel. Fourteen of the streams (12 inlets and the two outlets) were assessed to have fish habitat potential (*i.e.*, had consistent direction of flow and some degree of channelization). Aerial and ground surveys were conducted to map the habitat in the 14 streams. In June and July 1999, kick-sampling for eggs and aerial surveys for fish presence were conducted. Appendix IX.9 contains details on the methodology used in habitat mapping.

9.5.1.3 Summary of Results for Snap Lake

9.5.1.3.1 Phytoplankton

Phytoplankton, Chlorophyll* a *and nutrient concentrations were moderately low in Snap Lake

The phytoplankton, chl *a*, and nutrient concentrations in Snap Lake were moderately low (Table 9.5-2). This is typical of an oligo-mesotrophic lake that is characterized by low to moderate productivity with total phosphorus concentrations of 0.005 to 0.010 milligrams per litre (mg/L). The results for Snap Lake are consistent with similar lakes in the Slave Geological Province (Puznicki 1996). Chl *a* ranged from 0.44 micrograms per litre (µg/L) to 1.15 µg/L July, with average concentrations ranging from 0.72 µg/L (September) to 0.93 µg/L (July). Chl *a* at water quality sites WQ1 and WQ2 peaked in August, while values at WQ3 reached a maximum in July. Total phosphorus (TP) ranged form 0.005 to 0.014 mg/L and total kjeldahl nitrogen (TKN) concentrations ranged from 0.18 to 0.88 mg/L.

Table 9.5-2 Snap Lake Total Phosphorus, Total Kjeldahl Nitrogen and Chlorophyll *a* Concentrations in 1999

Sampling Station	July			August			September		
	Chl *a* (µg/L)[1]	TP (mg/L)	TKN (mg/L)	Chl *a* (µg/L)	TP (mg/L)	TKN (mg/L)	Chl *a* (µg/L)	TP (mg/L)	TKN (mg/L)
WQ1[2]	0.83	0.014	0.26	0.84	0.010	0.24	0.80	0.009	0.23
WQ3	0.81	0.005	0.27	0.93	0.009	0.18	0.44	0.006	0.27
WQ7	1.15	0.010	0.88	0.91	0.005	0.27	0.91	0.014	0.20
Average	0.93	0.01	0.47	0.89	0.01	0.23	0.72	0.01	0.23

1 Units: µg/L = microgram per litre; mg/L = milligram per litre.

2 Water Quality (WQ) stations 1, 3, and 7 (Figure 9.4-2).

Chl *a* = chlorophyll *a*; TP = total phosphorus; TKN = total kjeldahl nitrogen.

The average phytoplankton biomass was highest in September, with monthly variations in dominant taxa

The total average phytoplanktonic biomass was highest in September (444 milligram per cubic metre [mg/m^3]); monthly variations in the community structure were observed. In July, *Gymnodinium/Amphidinium* species were dominant at WQ3 (32%) and WQ7 (14%), while *Tabellaria flocculosa* accounted for 55% of the planktonic biomass at WQ1. In August, dinoflagellates were the most common taxonomic group at WQ1 (51%), while diatoms dominated the phytoplankton biomass at WQ3 (70%) and WQ7 (66%). The most common taxa at the three stations (WQ1, WQ3 and WQ7) in August were *Gymnodinium paradoxum*, *Cyclotella bodanica,* and *Tabellaria flocculosa*, respectively. In September, the phytoplanktonic biomass was predominantly comprised of diatoms (47-70%) at all three stations. *Cyclotella bodanica* was the most common species at WQ3 (17%), while *Tabellaria flocculosa* was dominant at WQ1 (34%) and WQ7 (28%). Detailed results from the phytoplankton surveys are presented in Appendix IX.10, Tables IX.10-1 and IX.10-2.

9.5.1.3.2 Zooplankton

The zooplankton community structure varied monthly with the average maximum biomass occurring in July

Zooplankton community structure varied over time, with the maximum average biomass occurring in July (13,8146.4 micrograms per cubic metre [µg/m^3]). In the shallow water areas, rotifers comprised 65% to 83% of the total number of zooplankton present in July, but Calanoida (*Heterocope septentrionalis, Leptodiaptomus sicilis, Leptodiaptomus minutus, Calanoid copepodid*) were the most dominant group by biomass. In July, *Laptodiaptomus minutus* accounted for 41% to 60% of the zooplankton biomass at the open water sites. In August and September, *Leptodiaptomus sicilis* dominated the open water sites (46% to 80% by biomass). The complete listing of zooplankton results can be found in Appendix IX.10, Tables IX.10-5 through IX.10-7.

Zooplankton ash free dry weight increased over the open water season reaching a maximum in September

Zooplankton ash-free dry weight ranged from 0.5 to 27.5 grams per square metre (g/m^2). Overall the ash-free dry weight increased over the open water season and reached a maximum in September at all three sampling sites (Table 9.5-3).

Table 9.5-3 Zooplankton Biomass in Snap Lake in 1999

Site	July		August		September	
	Dry Weight (g/m^2)[1]	Ash Free Dry Weight (g/m^2)	Dry Weight (g/m^2)	Ash Free Dry Weight (g/m^2)	Dry Weight (g/m^2)	Ash Free Dry Weight (g/m^2)
Shallow Water						
SH1[2]	15.5	12.6	-	-	-	-
SH2	17	12.7	-	-	-	-
SH3	4.6	3	-	-	-	-
Open Water						
WQ1	3.7	1.4	3.7	0.5	28.3	27.5
WQ3	2.3	0.5	4.8	1.7	7.2	5.6
WQ7	2.9	0.5	4.1	1.5	6.8	5.1

[1] g/m^2 = gram per square metre.

[2] Shallow habitat (SH) locations 1, 2, and 3; water quality (WQ) sites 1, 3, and 7.

- = not available

9.5.1.3.3 Benthic Invertebrates

The benthic Invertebrate community in Snap Lake was dominated by dipterans and nematodes

Benthic invertebrates were collected at four sites in Snap Lake during the fall sampling program in 1999. Mean total benthic invertebrate abundance varied between 5,000 and 7,400 organisms/square metre (m^2) in Snap Lake (see Table 9.5-4). Total taxonomic richness (the number of taxa at the lowest level of identification) varied little among sites. The total number of taxa found (calculated by pooling all the replicate samples at a site) was between 27 to 30. Chironomid midge larvae (predominantly tribes Chironomini and Tanytarsini) and nematode worms dominated the benthic community, accounting for 71% and 24%, respectively, of the total invertebrates collected in Snap Lake. Detailed benthic invertebrate sampling results are presented in Appendix IX.10, Tables IX.10-11 through IX.10-15.

Table 9.5-4 Benthic Invertebrate Data Collected in Snap Lake, Fall 1999

Variable	Snap Lake			
	SH1[1]	SH2	SH3	WQ3
Abundance and Taxonomic Richness (site mean ± standard error)				
Total abundance (no./m^2)[2]	7367 ± 2607	5010 ± 1230	5447 ± 919	6937 ± 1913
Mean richness/site	15.7 ± 1.6	14.5 ± 1.5	14.2 ± 1.3	11.8 ± 2.2
Total richness/site	30	27	28	27
Community Composition (site mean)				
Chironomidae	53.1	81.3	82.7	66.7
Nematoda	40.1	14.8	12.8	30.0
Mollusca	4.4	2.9	3.5	2.5
Other groups[3]	2.4	1.1	0.9	0.9

[1] Shallow habitat (SH) locations 1, 2, and 3; water quality (WQ) location 3.
[2] no./m^2 = number per square metre.
[3] Includes Oligochaeta, Hirudinea, Amphipoda, Hydracarina, Collembola, Ephemeroptera, Hemiptera, and Trichoptera.

9.5.1.3.4 Fish Inventory

Five species of fish were captured from Snap Lake in 1998, while seven species were captured in 1999

In 1998, five species of fish were gill-netted from Snap Lake including longnose sucker (*Catostomus catostomus*), burbot (*Lota lota*), lake trout (*Salvelinus namaycush*), round whitefish (*Prosopium cylindraceum*) and Arctic grayling (*Thymallus arcticus*) (Table 9.5-5). In 1999, seven species of fish were captured in Snap Lake. In addition to the species captured in 1998, lake chub (*Couesius plumbeus*) and slimy sculpin (*Cottus cognatus*) were captured in 1999 (Table 9.5-6).

Table 9.5-5 Summary of Fish Captured in Snap Lake, 1998[1]

Fish Species	Minnow Trap Captures	Electrofishing Captures	Gill Net Captures	Total
Longnose sucker	3	2	12	17
Burbot	2	1	1	4
Lake trout	1	0	29	30
Round whitefish	37	3	12	52
Arctic grayling	0	0	16	16

[1] Source: Hallam Knight Piesold Ltd. (1998).

Table 9.5-6 Summary of Fish Species Captured in Snap Lake, 1999

Waterbody	Fish Species Captured	Number Captured
Snap Lake	round whitefish	46
	Arctic grayling	10
	burbot	1
	lake chub	193
	longnose sucker	6
	lake trout	119
	slimy sculpin	1

Data regarding the characteristics of fish caught are summarized in Appendix IX.11

Length and weight data for fish that were caught in gill nets or angled are presented in Appendix IX.11, Table IX.11-2. The minimum, maximum, and mean fork length (millimetres [mm]), weight (grams [g]), age, gonad weight (g), condition factor, liver somatic index, and gonadal somatic index from fish caught in gillnets or angled are summarized in Appendix IX.11, Table IX.11-3. Appendix IX.11, Table IX.11-2 includes data on fish that were captured and released, and fish that were retained for tissue analysis. Common and scientific names of fish species are listed in Table IX.11-1 of Appendix IX.11. Definitions and details on how indices are calculated may be found in Appendix IX.9, Golder Associates Ltd. Technical Procedure 8.16.

9.5.1.3.5 Fish Tissue Analysis

Data regarding the concentration of metals in lake trout and round whitefish are presented in Tables IX.11-44 to IX.11-8

Muscle and liver tissue were collected from Snap Lake, the reference lake and MacKay Lake for metals analysis. Detailed results of tissue metal concentrations for flesh and liver are presented in Appendix IX.11, Table IX.11-4. Table IX.11-4 lists metal concentrations in lake trout flesh and liver from all three lakes. Appendix IX.11, Tables IX.11-5 through IX.11-8 detail round whitefish liver metal concentrations from all three lakes.

Bio-concentration factors for these fish species are also presented in Appendix IX.11-9 and 11-10

In addition to baseline metals concentrations, bio-concentration factors (BCF) were calculated for the parameters analyzed. The BCF is calculated by comparing baseline concentration of metal in the water from each lake to the tissue concentration. BCF values for lake trout and round whitefish from each of the three lakes are presented in Appendix IX.11, Tables IX.11-9 and IX.11-10 in.

9.5.1.3.6 Bathymetry

Snap Lake bathymetry map is shown in Figure 9.5-5

The bathymetry of Snap Lake is illustrated on Figure 9.5-5. Snap Lake has an average depth of 5 m. There are two deep areas in Snap Lake located at the extreme west end of the lake (45 m) and the southeast tip of the northwest peninsula (24 m).

9.5.1.3.7 Fish Habitat

Snap Lake habitat is composed of rocky shorelines, deep water areas with organic substrate

Snap Lake is a headwater lake with inflows from a number of small tributaries along the entire periphery of the lake. The shoreline is a mixture of boulder, exposed bedrock, and intermittent cobble. The shorelines of bays and sheltered areas are composed of organic matter and bedrock. The gradients of most near-shore areas are steep and lead to deeper water. Deep water substrates are composed of a thick layer of loose organic matter. The division between shoreline and deep water substrates was observed at the 3 m to 4 m depth contours (discussed below). The lake is characterized by numerous islands, rocky outcrops, and shoals.

Open tundra is the dominant shoreline habitat surrounding Snap Lake

The terrestrial shoreline of Snap Lake is dominated by open tundra (78%) with variable slope (Table 9.5-7, Figure 9.5-6). Other sections of terrestrial shoreline consist of coniferous forest (9%), sedge wetlands (<2%), and marsh (<1%). The slope of the terrain surrounding the lake varies from shallow to steep (Table 9.5-7).

Tables 9.5-8 and 9.5-9 summarize the key habitat components in Snap Lake

Habitat within Snap Lake was divided into three key components: nearshore habitat (shoreline to the 4 m depth contour), open water (>4 m in depth), and shoals (Table 9.5-8 and 9.5-9). Habitat types are discussed in detail in Appendix IX.12.

Boulder cobble substrate is the most common shoreline aquatic habitat in Snap Lake

Once the three main components were identified, aquatic habitat mapping of Snap Lake focused on the nearshore habitat (lakebed extending from the shore of the mainland and in-lake islands below the waterline) (Figure 9.5-6). The survey resulted in 2,135,146 m^2 of subsurface nearshore habitat being characterized to the 4 m contour line. The dominant nearshore habitat is boulder-cobble substrate comprising 1,573,011.7 m^2 (73.7%), followed by bedrock and bedrock-boulder (9%), boulder (6%), and bedrock-cobble (<1%). Inundated vegetation and inundated vegetation-boulder comprises less than 1% of the aquatic nearshore habitat. An additional 1% of the nearshore habitat is unknown or unrecorded (21,760 m^2). Descriptions of the habitat types observed in Snap Lake are provided in Table 9.5-9.

Figure 9.5-5 Snap Lake Bathymetry Map

Figure 9.5-6 Snap Lake Shoreline, Nearshore, and Shoal Habitat Map

Table 9.5-7 Summary of Shoreline Habitat and Slope Type for Snap Lake

Slope of Shoreline	Terrestrial Habitat Type[1]	Total Length of Shoreline Habitat (m)	Percentage of Total Shoreline Occupied by Habitat (%)
Shallow	coniferous forest	1,987	3
	coniferous forest/sedge wetland	809	1
	marsh	188	<1
	open tundra	20,965	27
	sedge wetland	157	<1
Moderate slope	coniferous forest	3,015	3
	coniferous forest/sedge wetland	499	<1
	open tundra	24,785	32
	open tundra/coniferous forest	199	<1
	sedge wetland	381	<1
Moderately steep	coniferous forest	1,838	2
	coniferous forest/sedge wetland	143	<1
	open tundra	8,096	10
	sedge wetland	763	<1
Steep	coniferous forest	256	<1
	open tundra	7,005	9
Unknown	unknown	7,454	9

[1] As defined in Section 10.3.

m = metre; % = percent.

Table 9.5-8 Summary of Major Aquatic Habitat Areas in Snap Lake

Habitat	Habitat #	Area of Major Habitat Types in Snap Lake (ha)	% of Total Area of each Habitat Type	Area of Major Habitat Types along the Northwest Peninsula (ha)	% of Total Area of Northwest Peninsula
Nearshore Habitat (waters edge to 4-m contour)					
Bo/Co	1	290.48	76.6	52.40	89.3
Bd	2	28.96	7.6	0	0
Bd/Bo	3	27.20	7.2	3.79	6.5
Bo	4	23.85	6.3	2.16	3.7
IV/Bo	5	4.24	1.1	0.34	0.6
unknown	-	2.18	0.6	0	0
Bd/Co	6	1.25	0.3	0	0
IV	7	1.21	0.3	0	0
Total Nearshore Habitat		379.37	100	58.69	100
Shoal Habitat (primary and secondary lake trout spawning areas to the 6 m contour)					
Primary	9	16.82	50	-	-
Secondary	10	16.34	50	-	-
Total Shoal Habitat		33.16	100	-	-
Deep Water (>4 m in depth)					
Deep water	8	276.36	100	-	-
Open Water (entire lake area)					
Open water	-	688.89	-	-	-

Note: m = metre; ha = hectare; % = percent; dash line (-) = not applicable; unknown = not classified; Bd = Bedrock, Bo = boulder (>25 cm), C = cobble (>6.5 cm), R = rubble (>6.5 cm, angular), G = gravel (>0.2 cm), S = sand (>0.06 mm) and CS = clay/silt (<0.06 mm).

Table 9.5-9 Snap Lake Fish Habitat Descriptions

Habitat Type	Description
Shorelines (< 4 m)	
Boulder/cobble (Bo/Co)	First 2 m steep gradient. Rocks relatively clean of algae and silt. Substrates 70% boulder/30% cobble on average. Gradient shallow after 2 m. Substrates 40% boulder: 40% silt or organic matter: 10% cobble. Rocky substrates covered in algae/silt and not more than one layer thick.
Bedrock (Bd)	Flat or shallow gradient sheet of rock. Surface clean near shore but progressively covered with algae/silt with depth.
Bedrock/boulder (Bd/Bo)	Flat or shallow gradient sheet of rock. Boulders scattered on surface of bedrock. No more than one layer thick of boulders at any location. Surface of all rocky substrates clean near shore but progressively covered with algae/silt with depth.
Boulder (Bo)	First 2 m steep gradient. Rocks relatively clean of algae and silt. Substrates 100% boulder. Gradient shallow after 2 m. Substrates 50% boulder: 50% silt or organic matter. Rocky substrates covered in algae/silt and not more than one layer thick.
Inundated vegetation/ boulder (IV/Bo)	Occurs only in sheltered bays or at the mouth of streams. Gradient flat or very shallow. Substrates a varying mixture of scattered boulders with silt substrates in between. Emergent vegetation found in between boulders. Boulders covered by thick layer of algae and silt.
Bedrock/cobble (Bd/Co)	Flat or shallow gradient sheet of rock. Cobble scattered on surface of bedrock. No more than one layer thick of cobble at any location. Surface of all rocky substrates clean near shore but progressively covered with algae/silt with depth.
Inundated vegetation (IV)	Occurs only in sheltered bays or at the mouth of streams. Gradient flat or very shallow. Substrates silt and organic matter. Emergent vegetation scattered throughout area.
Shoals (0 – 6 m)	
Primary	Mixture of boulder (60-70%) and cobble (40-30%) to a depth of 6 m. Gradient steep to moderate. Rocky substrates clean near surface but become progressively covered with silt and algae with depth. Boulder/cobble layer two to three rocks deep in most areas.
Secondary	Mixture of boulder (80-90%) and cobble (20-10%) to a depth of 6 m. Gradient shallow to moderate. Rocky substrates clean near surface but become progressively covered with silt and algae with depth. Algae/silt cover heavy in areas with low gradient. Boulder/cobble layer one to two rocks deep in most areas.
Deep Water (>4 m)	
	Substrates in deep water dominated by thick layer of diatoms and other loose organic matter. Surface layer of diatoms alive but lower layers exhibit in varying degrees of decay. Cover for forage fish or macroinvertebrates poor.

m = metre; % = percent.

9.5.1.3.8 Lake Trout Spawning Survey

The most active lake trout spawning ground was located near the centre of Snap Lake

The preferred lake trout spawning habitat in Snap Lake was boulder-bedrock shoals near deep water. The most appropriate rock was available between depths of 2 m and 6 m, and was fully exposed to wind and wave action. The most active spawning ground was located near the centre of Snap Lake (Figure 9.5-6) where 31,170 m^2 of primary spawning habitat was identified. In other regions of the lake, 40,582 m^2 of secondary spawning grounds were identified. All of the spawning grounds identified were located a considerable distance from the proposed Snap Lake Diamond Project site. No spawning was observed along the shoreline of the northwest peninsula or in the west arm of Snap Lake.

9.5.1.4 Summary of Results for Reference and Regional Lakes

9.5.1.4.1 Phytoplankton

Phytoplankton chlorophyll** a **and nutrient concentrations in the reference lake were moderately low

In the reference lake, phytoplankton chl *a* and nutrient concentrations were moderately low, which is typical of oligo-mesotrophic lakes (total phosphorus 0.005-0.010 mg/L) and is similar to lakes in the Slave Geologic Province (Puznicki 1996). Like Snap Lake, chl *a* levels peaked in August (average 0.94 µg/L), with the highest concentration (1.14 µg/L), occurring at reference sampling station WQR3 (Table 9.5-10). TP concentrations were relatively low, ranging from 0.004 to 0.014 mg/L. In August, TP concentrations were highest (average 0.013 mg/L), and tended towards meso-eutrophic status (0.010-0.030 mg/L). The average concentration of TP for all locations and all months was 0.01 mg/L, which is equal to the concentration in Snap Lake. TKN concentrations ranged from 0.20 to 0.90 mg/L.

Table 9.5-10 Reference Lake Total Phosphorus, Total Kjeldahl Nitrogen, and Chlorophyll *a* Concentrations in 1999

Sampling Station	July			August			September		
	Chl *a* (µg/L)[1]	TP (mg/L)[2]	TKN (mg/L)	Chl *a* (µg/L)	TP (mg/L)	TKN (mg/L)	Chl *a* (µg/L)	TP (mg/L)	TKN (mg/L)
WQR1[3]	0.79	0.004	0.40	0.84	0.012	0.2	0.76	0.005	0.24
WQR3	0.68	0.01	0.90	1.14	0.014	0.2	0.77	0.006	0.21
WQR7	0.98	0.01	0.89	0.83	0.014	0.2	0.7	0.008	0.21
Average	0.81	0.01	0.65	0.94	0.01	0.20	0.74	0.01	0.23

[1] µg/L = microgram per litre.
[2] mg/L = milligram per litre.
[3] WQR = Water Quality Reference (WQR) lake station.

The highest average phytoplanktonic biomass occurred in July and September, with monthly variations in dominant taxa

The highest average phytoplanktonic biomass occurred in July (855 mg/m^3) and September (845 mg/m^3), with monthly variations in dominant taxa. In July, the samples from WQR3 (1,118 mg/m^3) and WQR7 (731 mg/m^3) had the highest biomass of phytoplankton. The largest biomass of plankton in September occurred at WQR3 (1,266 mg/m^3). In July, *Gymnodinium paradoxum* (golden brown algae) accounted for 56% of the phytoplankton biomass at WQR1, while *Ochromonas* species were the most abundant taxa at WQR2 (65%) and WQR3 (50%). Dinoflagellates accounted for most of the phytoplankton biomass in August (40% to 61%). *Gymnodinium paradoxum* was the most abundant taxa at WQR1 (56%) and WQR3 (20%), and *Ceratium hirundinella* was dominant at WQR2 (44%). In September, the diatom *Asterionella formosa* was the most abundant species at all three sites (25% to 50%). Although different in phytoplankton community structure than Snap Lake, the phytoplankton community composition is still typical of nutrient-poor Arctic lakes (Wetzel 1983). The highest average phytoplanktonic biomass occurred in July (855 mg/m^3) and September (845 mg/m^3). There were monthly variations in dominant taxa; the density was greatest in September (488,886/L). Detailed results of the phytoplankton survey in the reference lake are provided in Appendix IX.10, Tables IX.10-3 and IX.10-4.

9.5.1.4.2 Zooplankton

Reference lake zooplankton ash free dry weight increased over the open water season

In the reference lake, zooplankton ash-free dry weight ranged from 0.6 to 10.9 g/m^2. Over the course of the open water season the mass of zooplankton in the reference lake increased and reached an average maximum of 7.3 g/m^2 in September (Table 9.5-11).

Table 9.5-11 Reference Lake Zooplankton Biomass in 1999

Site	July		August		September	
	Dry Weight (g/m^2)[1]	Ash Free Dry Weight (g/m^2)	Dry Weight (g/m^2)	Ash Free Dry Weight (g/m^2)	Dry Weight (g/m^2)	Ash Free Dry Weight (g/m^2)
WQR1[2]	2.4	0.6	4.4	1.4	13.1	10.9
WQR3	4.8	2.3	4.7	2	5.3	4
WQR7	6.2	3.8	3.9	2.2	8	7

1 g/m^2 = grams per square metre.
2 WQR 1,3,7 = water quality reference (WQR) sampling station.

Zooplankton community structure varied over time, with the maximum average biomass occurring in July

Zooplankton community structure varied over time, with the maximum average biomass occurring in July (53,163 μg/m^3). In the shallow habitat reference sample SHR1 collected in July, *Daphnia longiremis* and *Bosmina longirostris* each accounted for 24% of the total phytoplanktonic biomass. *Conochilus unicornis* and *Leptodiaptomus sicilis* were the most common taxa at SHR2 (26%) and SHR3 (33%) in July. *Diacyclops bicuspidatus* was the most abundant species in the open water sites of WQR1 (18%), WQR3 (36%), and WQR7 (40%) in July. *Diacyclops bicuspidatus* continued to dominate the zooplankton biomass at WQR3 and WQR7 in August. *Diacyclops bicuspidatus* (30%) and *Holopedium gibberum* (33%) co-dominated the zooplankton at WQR1 in August. In September, *Diacyclops bicuspidatus* was the most abundant taxa at WQR1 (41%) and WQR7 (51%), while *Holopedium gibberum* dominated at WQR3 (46%).

Zooplankton survey results for the reference lake are provided in Appendix IX.10, Tables IX.10-8 and IX.10-9.

9.5.1.4.3 Benthic Invertebrate Community

Reference Lake benthic invertebrate community was dominated by dipterans and nematodes

Benthic invertebrates were collected at four sites in the reference lake during the fall sampling program in 1999. Mean total benthic invertebrate abundance varied between 1,900 and 34,600 organisms/m^2 in the reference lake (Table 9.5-12). Total abundance was highly variable at two sites in the reference lake (WQR3 and SHR2). Taxonomic richness varied little among sites or lakes. The total number of taxa was between 27 and 31 in the reference lake. As in Snap Lake, chironomid midge larvae and nematode worms dominated the benthic community, accounting for 62% and 33%, respectively, of the invertebrates collected. Detailed results of the benthic invertebrate community survey in the reference lake are provided in Appendix IX.10, Tables IX.10-11 through IX.15.

Table 9.5-12 Benthic Invertebrate Data Collected in the Reference Lake, Fall 1999

Variable	WQR1[1]	WQR3	WQR7	SHR2
Abundance and Taxonomic Richness (site mean ± standard error)				
Total abundance (no./m^2)[2]	9,231 ± 1741	23,994 ± 5760	1,935 ± 271	34,601 ± 8711
Mean richness/site	12.5 ± 0.8	11.2 ± 0.8	10.7 ± 1.0	10.7 ± 2.1
Total richness/site	31	27	28	28
Community Composition (site mean)				
Chironomidae	47.2	68.3	55.2	78.1
Nematoda	51.2	30.1	31.1	20.4
Mollusca	1.1	1.4	6.4	1.2
Other groups[3]	0.5	0.2	7.3	0.3

[1] WQR (water quality reference) lake sample site; SHR (shallow habitat reference) lake sample site.
[2] no./m^2 = number per square metre.
[3] Includes Oligochaeta, Hirudinea, Amphipoda, Hydracarina, Collembola, Ephemeroptera, Hemiptera, and Trichoptera.

9.5.1.4.4 Fish Inventory

Four species of fish were captured from the reference lake in 1999 and three species were captured in Mackay Lake in 2001

Four species of fish were captured in the reference lake in 1999. As in Snap Lake, lake trout, longnose sucker, and round whitefish were abundant. No burbot were captured in the reference lake (Table 9.5-13). In 2001, the fish inventory at MacKay Lake resulted in the capture of lake trout, round whitefish, and one Arctic grayling (Appendix IX.11). Common and scientific names of fish species are also listed in Appendix IX.11, Table IX.11-1.

Table 9.5-13 Fish Species Captured in the Reference Lake in 1999 and MacKay Lake in 2001

Waterbody	Fish Species Captured	# Captured
Reference Lake	lake trout	39
	round whitefish	24
	longnose sucker	19
	brook stickleback	5
MacKay Lake	lake trout	24
	round whitefish	36
	Arctic grayling	1

Data regarding the characteristics of fish caught are summarized in Appendix IX.11

Length and weight data for fish that were caught in gill nets or angled are presented in Appendix IX.11, Table IX.11-2. Table IX.11-3 (Appendix IX.11) summarizes the minimum, maximum and mean fork length (mm), weight (g), age, gonad weight (g), condition factor, liver somatic index, and gonadal somatic index from fish caught in gillnets or angled. Table IX.11-3 includes data on fish that were captured and released and fish that were retained for tissue analysis. Definitions and details on how indices are calculated may be found in Appendix IX.9.

9.5.1.4.5 Fish Tissue Analysis

Data regarding the concentration of metals in lake trout and round whitefish are presented in Appendix IX.11

Muscle and liver tissue were collected from the reference lake and MacKay Lake for metals analysis. Data on metal concentrations in fish tissues are presented in Appendix IX.11. Appendix IX.11, Table IX.11-4 lists metal concentrations in lake trout flesh and liver from both lakes. Appendix IX.11, Tables IX.11-5 through IX.11-8 list metal concentrations in the flesh and liver of round whitefish from both lakes. Baseline BCFs were also calculated for both fish species in these lakes and are found in Appendix IX.11, Tables IX.11-9 and IX.11-10.

9.5.1.5 Inland Lakes and Streams

9.5.1.5.1 Bathymetry

Bathymetry maps for the inland lakes near the site and lakes to the north are provided

The bathymetry of IL 2 to 9 and NL 1 to 4 is detailed in Figures 9.5-7 through 9.5-9. The inland lakes range in depth from 0.8 m (IL4) to greater than 4 m (IL5). Lake areas range from 0.53 ha (IL 4) to 8.2 ha (IL 5). The lakes to the north were larger and deeper than the inland lakes around the Snap Lake Diamond Project footprint. Depth ranged from 3 m (NL2 and 4) to 7 m (NL3). In area, NL3 was the smallest (5.29 ha) and NL1 was the largest (14.05 ha).

Figure 9.5-7 Bathymetry Map of Lakes IL2, IL3, IL4 and IL5

Figure 9.5-8 Bathymetry Map of Lakes IL6, IL7, IL8 and IL9

Figure 9.5-9 Bathymetry Map of Lakes NL1, NL2, NL3 and NL4

9.5.1.5.2 Fish Inventory

Of the lakes located near the proposed project footprint, fish were only captured in IL5, all four of the NLs were fish bearing in 1999

Of the inland lakes near the site, fish were only captured in IL5 and were observed in IL3 but not captured. All four of the lakes to the north were fish bearing. Lake chub was the only species captured or observed in the inland lakes. Lake chub were captured in all the lakes to the north except NL3. Longnose sucker were captured in NL2 and NL3 (Table 9.5-14).

Table 9.5-14 Summary of Fish Species Captured in Small Lakes, 1999 and 2001

Waterbody	Fish Species Captured	# Captured
IL5	lake chub	18
NL1	lake chub	21
NL2	lake chub	67
	longnose sucker	10
NL3	longnose sucker	6
NL4	lake chub	17

9.5.1.5.3 Fish Habitat

The inland lakes near the site had little or no shoreline vegetation cover and mainly rock substrates; few provided overwintering habitat

The small lakes located near the proposed project footprint in the Snap Lake watershed and the lakes located to the north outside of the Snap Lake watershed are typical of small lakes in the southern Arctic region above the tree line. Little or no overhanging cover from shoreline vegetation was observed around the lakes, except where tributary streams were present. In these areas, trees and shrubs were present. The shoreline was mainly boulder with a variety of secondary substrates including cobble and bedrock (Table 9.5-15). Substrates in bays and isolated areas were composed of a mixture of boulders and organic matter. Most lakes investigated had a single, centrally located deep basin although several were shallow in all areas (Figures 9.5-7 to 9.5-9). Only IL5 and IL7 were found to have maximum depths >3 m, the depth generally required to provide overwintering habitat with high enough oxygen concentrations for fish species to survive. Of these two lakes, only IL5 was found to have fish present.

Table 9.5-15 Summary of 1999 Lake Habitat and Fish Capture Investigations

Lake	Area (ha)	Maximum Depth (m)	Connection to Other Lakes via Streams	Basin Characteristics	Fishing Gear Used	Fish Catch	Proximity to Project Footprint	Habitat Assessment
IL2	1.98	2.1	Ephemeral	Dominated by shallow <1 m shoals Small >1 m basin	Gill nets/ minnow traps	None	Near airstrip	Marginally suitable small bodied fish habitat, exhibits some capability to overwinter small fish
IL3	1.15	1.9	Ephemeral (no defined channel)	Dominated by shallow <1 m areas, organic/detritus substrate small > 1 m basin	Gill nets/ minnow traps	None (loon observed on lake)	Near airstrip	Marginally suitable small bodied fish habitat, exhibits some capability to overwinter small fish
IL4	0.53	0.8	Ephemeral (no defined channel)	All shallow water, mean depth 0.6 m	None used due to lack of depth	n/a	Near airstrip	No overwintering habitat for any fish species
IL5	8.21	4.0	Defined channel with suitable habitat for cyprinids	Shoreline dominated by large boulders, some macrophytes present, some areas of fine organic sediments present	Gill nets/ minnow traps	18 lake chub	Near airstrip	Suitable small bodied fish habitat, overwintering available for these species
IL6	2.88	2.5	Ephemeral connection to IL7 through bog area Ephemeral flow to Snap Lake through vegetated terrain, no visible stream channel	Shoreline dominated by vertical fractured bedrock, lake substrate dominated by boulder and cobble with small areas of fine organic substrates	Gill nets/ minnow traps	none	Near north pile	Marginally suitable small bodied fish habitat, exhibits some capability to overwinter small fish
IL7	2.4	3.5	Ephemeral narrow stream between this lake and IL6 through bog, no flow detected July 2001	Shoreline dominated by shrubs and grasses interspersed with boulders and spruce trees. Steep drop-off surrounds entire lake, substrate dominated by boulder cobble and silt.	Gill nets/ minnow traps	None	Within north pile foot print, partially covered by north pile, to be used as a sedimentation pond	Marginally suitable small bodied fish habitat, exhibits some capability to overwinter small fish

Table 9.5-15 Summary of 1999 Lake Habitat and Fish Capture Investigations (continued)

Lake	Area (ha)	Maximum Depth (m)	Connection to Other Lakes via Streams	Basin Characteristics	Fishing Gear Used	Fish Catch	Proximity to Project Footprint	Habitat Assessment
IL8	6.5	1.1	None	Shoreline dominated by shrubs interspersed with boulders and spruce trees; 0.5 m bedrock drop-off surrounds entire lake, substrate dominated by boulder cobble interspersed with silt	Gill nets/ minnow traps	None	Near north pile footprint, to be used as a sedimentation pond	No overwintering habitat for any fish species
IL9	0.8	1.6	Ephemeral channel present between bog to the east of the lake and IL9, no flow detected in July 2001	Shoreline muskeg and grasses, some macrophytes present, some areas of fine organic sediments present	Gill nets/ minnow traps	none	Near north pile footprint, to be used as a sedimentation pond	No overwintering habitat for any fish species
NL1	14.05	5	No data	Two basin lake with shoreline dominated by vertical fractured bedrock, lake substrate dominated by boulder and fines	Gill nets/ minnow traps	21 lake chub	Not in Snap Lake Diamond Project footprint, located outside of Snap Lake watershed	Suitable small bodied fish habitat, overwintering available for these species
NL2	12.03	3	No data	Dominated by shallow water with boulder slopes	Gill nets/ minnow traps	67 lake chub 10 longnose sucker	Not in Snap Lake Diamond Project footprint, located outside of Snap Lake watershed	Suitable small bodied fish habitat, overwintering available for these species
NL3	5.29	6	Emergent vegetation in stream channel	Moderately deep basin in centre of lake, boulder-bedrock shoreline	Gill nets/ minnow traps	6 longnose sucker	Not in Snap Lake Diamond Project footprint, located outside of Snap Lake watershed	Suitable small bodied fish habitat, limited overwintering available for these species
NL4	7.56	3	No data	Shallow throughout most of lake, boulder-bedrock shoreline	Gill nets/ minnow traps	17 lake chub	Not in Snap Lake Diamond Project footprint, located outside of Snap Lake watershed	Suitable small bodied fish habitat, limited overwintering habitat available

9.5.1.5.4 Stream Surveys

During the early June 1999 survey, fish were observed either in or near the mouth of seven streams

Thirty streams were identified in the Snap Lake watershed (Figure 9.5-3). An additional stream (S31) was identified near the project. Fourteen of the streams around Snap Lake were identified with fish habitat potential, twelve inlet streams and two outlet streams. Fish were observed either in or near the mouth of seven of the 14 streams (S1, S4, S20, S25, S27, H1, and H2) during the 1999 surveys. The habitat survey information collected for the streams investigated in 1999 is presented in Table 9.5-16. This includes a summary of the habitat investigations and summaries of fish observations (including egg collections) in the streams. Definitions for terms used to describe fish habitat are detailed in Appendix IX.9, Technical Procedure 8.5-1.

Four streams were identified in the project footprint

Streams S28, S29, S30, and S31 are located within the footprint of the proposed Snap Lake Diamond Project. The flow regime of S30 was altered with the construction of the PKC during the AEP. The airstrip built for the AEP crossed S30. Habitat information on S31 and S30 were presented in documentation related to the AEP and will be only briefly mentioned here. S28 and S29 (Table 9.5-17) did not have defined stream channels and can be described as runoff areas with ephemeral flow through vegetated terrain. The headwater of S29 is presumed to be IL6 and the headwater of S28 is presumed to be IL 8 (Figure 9.5-3), but no defined connection to the lakes were observed.

9.5.1.6 Ecological Process in Local Study Area Lakes

9.5.1.6.1 Chemical Variables

Winter thermal stratification and bioavailability affect exposure to chemicals

The chemical transport mechanisms in Snap Lake will concentrate toxicants near the sediment during the winter. Sediment binding and complexation will decrease the bioavailability of metals.

Chemical Transport

In the winter, sub-arctic and Arctic lakes exhibit thermal inversion

During the fall, surface water is rapidly cooled by northern winds. This epilimnetic cooling progresses at a faster rate than the hypolimnion. Below 4^0C (maximum density of water), a thermocline becomes established and turbulent, full column, mixing is reduced due to the density gradient. By the time ice-over occurs a strong inverse stratification (cooler, less dense water on top, warmer, more dense water on bottom) is established and often persists throughout the year. Ice-over in the RSA persists for up to nine months of the year.

Table 9.5- 16 Snap Lake Stream Survey Results for Streams with Fish Habitat Potential, 1999

Stream	Length (m)	Average Depth (m)	Dominant Habitat	Dominant Substrate	Fish Observed/Caught	Proximity to Project Footprint	Habitat Assessment
S1(WQ)	3,000	0.4 1.0 max in pools	Avg. width 1.9 m, run class 2 and 3 dominant <10% of stream riffles and pools	Boulder/bedrock dominant, some patches with cobble >30% silt present in pools	Early June, several unidentified 20 - 30 cm fish observed in 1 pool Several fish also observed at mouth of stream	None	Suitable fish habitat for small fish Suitable spawning habitat based on other available habitat in area Still flowing in July slightly, thus has potential to be suitable rearing habitat
S2(WQ)	800	0.2	Avg. width 1.5 m		None	None	Unsuitable habitat due to indistinct stream mouth and ephemeral nature
S4	700	0.2 0.72 max in pools	Avg. width 2.5 m Run class 2 and 3 dominant	Boulder/bedrock dominant	None in early June June 25, 6 small fish observed, one captured briefly, identified as cyprinid species	None	Braided channel through boulder garden and willow thickets may act as barriers to fish Suitable habitat for feeding Ephemeral and shallow
S7(WQ)	250	0.2	Avg. width 1.4 m Run class 4 and boulder garden dominant No pools or riffles	Boulder and inundated vegetation dominant	None	None	Braided channel through boulder garden and willow thickets may act as barriers to fish Unsuitable habitat for most life stages. Ephemeral and shallow
S10(WQ)	700	0.4 >1 m max in pools	Avg. width 1.5 m Run class 3 through inundated vegetation and boulder gardens approx. 25% pools and flats	Boulder and inundated vegetation dominant	None	None	Downstream areas accessible to fish, unsuitable spawning habitat however Subsurface flow through boulder gardens blocking passage further upstream

Table 9.5-16 Snap Lake Stream Survey Results for Streams with Fish Habitat Potential, 1999 (continued)

Stream	Length (m)	Average Depth (m)	Dominant Habitat	Dominant Substrate	Fish Observed/Caught	Proximity to Project Footprint	Habitat Assessment
S12	75	0.75	Avg. width 5 m Run class 2	Boulder/bedrock dominant	None	None	Short channel between Snap Lake and an inland lake, no barriers to passage Marginal suitability for spawning and rearing
S20(WQ)	300	0.5 0.75 max in pools	Avg. width 5 m Run class 2 and 3 dominant 5% lateral pools present	Boulder dominant	None in early June Approx. 5 fish observed among rocks June 25th, 3 - 5 cm long, possibly juvenile Arctic grayling	None	Boulder gardens likely barriers to passage under low flow conditions Generally suitable habitat for rearing but marginal for spawning (poor substrate)
S22	100	0.5	Avg. width 1.0 m Run class 2 dominant No pools	Boulder with some small patches of cobble	None	None	Barrier to fish passage at mouth – chute
S24	125	0.25	Avg. width 2 m Run class 3 dominant No pools	Boulder dominant, one relatively large (10% of stream) area of gravel/cobble	Early June, unidentified fish observed at inland lake outlet	None	Short channel between Snap Lake and an inland lake, no barriers to passage Suitable for spawning but marginal for rearing
S25(WQ)	100	0.5	Avg. width 2 m Run class 4 and 3 dominant No pools	Boulder dominant with inundated vegetation	None in early June June 26, 10 small unidentified cyprinids observed (likely lake chub)	None	Barriers to fish passage – fall/chute at mouth and undefined channel through vegetation further upstream
S27(WQ)	150	0.4	Avg. width 2.25 m Run class 4 and 3 dominant No pools	Boulder dominant with inundated vegetation	None in early June June 26, several small eggs collected and 1 small unidentified fish observed	None	Braided channel through boulder garden and willow thickets may act as barriers to fish

Table 9.5-16 Snap Lake Stream Survey Results for Streams with Fish Habitat Potential, 1999 (continued)

Stream	Length (m)	Average Depth (m)	Dominant Habitat	Dominant Substrate	Fish Observed/Caught	Proximity to Project Footprint	Habitat Assessment
S30	500	0.4	Avg. width 2 m Run class 3 through boulder gardens dominant	Boulder dominant with inundated vegetation	None	PKC blocks flow source	Undefined channel through boulder garden and willow thickets, subsurface flow, small chute at mouth of stream all may act as barriers to fish; unsuitable habitat Ephemeral and shallow
S31	500	No surface flow.	Average width could not be recorded due to lack of defined channel and no surface flow	Boulder dominant with inundated shrubs and grasses.	None	Airstrip bisects stream; crossed with culvert	Ephemeral and shallow; dry in July
H1 (outlet from Snap Lake)	100	0.5	Avg. width 3 m Run class 3 and riffle dominant No pools	Boulder/bedrock dominant	None in early June 1 slimy sculpin June 25 1 unknown egg collected	None	Barriers to fish movement - Subsurface flow through boulder garden at outlet of Snap Lake Water fall over bedrock at inlet to next lake downstream
H2 (outlet from Snap Lake)	125	0.5	Avg. width 9 m Run class 2 and riffle dominant, boulder gardens at both ends of stream No pools	Boulder dominant	None in early June Late June: 20 cm juv. round whitefish captured June 25 in isolated pool 5 other 3-5 cm fish observed among rocks, possibly juv. Arctic grayling 12 unknown eggs collected by kick sampling near Snap Lake (upstream) end of stream	None	Upstream fish movement into Snap Lake unlikely due to velocity barrier Moderate spawning habitat present at Snap Lake end of stream, upstream of boulder gardens Moderate rearing habitat available

Table 9.5-17 Snap Lake Stream Survey Results Ephemeral Channels, 1999

Stream	Approximate Length[1]	Habitat Assessment	Proximity to Project Footprint
S3	486	ephemeral, no defined channel	none
S5	167	ephemeral, no defined channel	none
S6	284	ephemeral, no defined channel	none
S8	199	ephemeral, no defined channel	none
S9	161	ephemeral, no defined channel	none
S11	100	ephemeral, no defined channel	none
S13	57	ephemeral, no defined channel	none
S14	208	ephemeral, no defined channel	none
S15	195	ephemeral, no defined channel	none
S16	391	ephemeral, no defined channel	none
S17	171	ephemeral, no defined channel	none
S18	140	ephemeral, no defined channel	none
S19	100	ephemeral, no defined channel	none
S21	113	ephemeral, no defined channel	none
S23	89	ephemeral, no defined channel	none
S26	69	ephemeral, no defined channel	none
S28	328	ephemeral, no defined channel	near footprint, sub-basin P
S29	128	ephemeral, no defined channel	near north pile footprint, sub-basin O

[1] Based on 1:50 000 NTS map projection.

Water discharged to Snap Lake will be slightly warmer than the lake during the winter

The majority of chemicals will migrate to the lowest areas in Snap Lake because they are slightly warmer than the surface water (Section 9.4). Thus, the majority of toxicological exposure will be in a layer of water 8 m deep to the bottom during winter in Snap Lake.

Groundwater recharge enters the north and northeast lakes at the bottom

The groundwater will also be at a higher temperature than the overlying surface water during the winter months. Groundwater will enter the north and northeast lakes at the bottom and remain there. The majority of toxicological exposure will be in the bottom water column and sediment layer.

Bioavailability of Metals

It is expected that sediment metals will be sequestered and largely unavailable to biota

Elemental metals (*i.e.* Cr^{6+}, Al^{3+} *etc*.) are the primary form of toxic species (Mance 1990). Consistent with this theory, total metal concentrations in sediments do not correlate to biological affect (Chapman *et al.* 1998). This is largely due to metal binding and ligand complexation. The primary mechanisms that may lower metal availability in Snap Lake include:

- iron and manganese oxyhydroxide (FeOOH and MnOOH) binding; and,
- ligand complexation (OH^-, Cl^-, HCO_3^-, SO_4^{2-} and CO_3^{2-}).

High discharge of iron and manganese will contribute to greater binding

Baseline average Snap Lake iron and manganese concentrations are 0.03 µg/L and 3.4 µg/L respectively. The average discharge concentrations of iron and manganese are 448 µg/L and 30 µg/L respectively. Given the sediment chemical transport characteristics and elevated oxyhydroxide concentration, metal binding in Snap Lake sediments will be considerable.

High chloride and sulfate discharge concentrations will contribute to elevated ligand complexation

The baseline average Snap Lake chloride and sulfate concentrations are <0.2 mg/L and mg/L, respectively. Predicted chloride and sulfate concentrations in the mine water discharge are 237 mg/L and 17 mg/L, respectively. A considerable amount of metals will be bound to chloride and sulfate through ligand complexation. Other anionic complexes are expected to increase and contribute to decreased metal bioavailability but their discharge concentrations have not been predicted.

9.5.1.6.2 Ecological Variables

Ecology of organisms will affect exposure to chemicals

The sediment, pore/interstitial water and overlying water in all lakes are the most vulnerable to chemical exposure. Aquatic ecology of sub-arctic phytoplankton, zooplankton and benthos will play an important role in exposure to chemicals.

Phytoplankton

Phytoplankton positioning is governed by available light

During the winter in temperate zones, most phytoplankton species will migrate below the ice-layer to scavenge available light. As light transmission decreases, up to a point, photosynthetic efficiency increases. Snow cover, consolidated ice, and onset of the polar winter substantially alters available light and phytoplankton dynamics.

Most studies are done on sea ice

Most research concerning polar phytoplankton dynamics is centred around sea-ice studies. Few studies link sea-ice and polar lake dynamics. Thus, extrapolation, where possible, is made from sea ice studies.

During the polar winter, the most common open water phytoplankton are diatoms

Asterionella formosa and *Tabellaria* are the primary diatom genus/species in Snap Lake and the reference lake. *Asterionella formosa* is distinct to temperate and Arctic climates, as it is unable to survive in temperatures greater than 25^0C (Lampert and Sommer 1997) but will grow at temperatures between 0.5-24°C (Horne and Goldman 1994). *Asterionella*, like many diatoms, become resuspended during the spring melt and mixing (Wetzel 2001).

Diatoms adapt to polar conditions

Sea-ice diatoms adapt to polar conditions through several physiological mechanisms including (Palmisano and Sullivan 1983):

- decline in growth rate;
- reduction of photosynthesis;
- decrease in ATP; and,
- storage and use of carbohydrates.

Metabolic suppression will reduce biomass and production during the winter

Metabolic suppression contributes substantially to a reduction in winter biomass (Cockell *et al*. 2000). Phytoplankton biomass may decrease by up to two orders of magnitude. In the study by Clarke and Leakey (1996), biomass changed from 1-2 mg/m^3 (winter) to 200 mg/m^3 (summer). Kottmier and Sullivan (1987) demonstrated primary and secondary production decreases between unconsolidated and consolidated pack ice. Thus, expected production in Snap Lake during the winter would be further reduced when compared to many sea-ice studies.

Many Arctic diatoms do not have resting spores but settle to the bottom in winter

Most Arctic diatoms do not appear to have resting spores (Clarke and Leakey 1996). In fact, *Asterionella formosa* is classified as holoplanktonic (Lee 1993). Metabolic suppression appears to support a limited population of phytoplankton, mostly diatoms, over the winter. The diatoms generally settle to the bottom over winter according to their intrinsic settling rate (Lee 1993) and lack of metabolism required to remain buoyant.

Toxic effects to phytoplankton are increased due to sediment positioning and decreased by lowered metabolic rate

The expected phytoplankton dynamics in Snap Lake and the north lake will have mixed effects relating to production and toxicology. During most of the winter, when light availability and transmission is severely limited, phytoplankton will settle to the substrate. The substrate concentration of phytoplankton are a result of reduced mixing and inverse thermal stratification. Although the phytoplankton biomass will reside in the areas with the highest chemical concentrations, toxicological effects will be limited by reduced metabolism.

Zooplankton

Zooplankton display distinct ecological adaptations to sub-arctic and Arctic winters; copepods for example overwinter near the sediment

Copepods were the dominant zooplankton class in both Snap Lake and the reference lake. According to Cockell *et al*. (2000), many copepods overwinter at depth, migrating to the surface for the spring spawning. The new generation begins to descend to greater depths as the summer progresses until overwintering depths are reached and they spend winter as late copepodites. This pattern is similar in freshwater lakes (Stienhart and Wertsbaugh 1999). Chymko (1976) hypothesized that Cyclopoid copepods overwintered in a copodite IV resting stage in the sediments of Lesser Slave Lake. The overwintering resting stage is supported by fatty tissue analysis (Orcutt and Proter 1983 *et al*. 1983, Cockell *et al*. 2000), storage of lipids for overwintering with dormancy, and reductions in metabolic rate with combustion of body lipids.

The reduction in metabolic rate is a result of phytoplankton food chain effects

Herbivorous zooplankton are not able to access phytoplankton in the later winter months and carnivorous zooplankton are affected by the reduction in prey. This is one reason for the low secondary productivity and reduced biodiversity of zooplankton in Arctic lakes (Christoffersen 2002).

Zooplankton will likely reside along the sediments

The overwintering Snap Lake and north lake zooplankton assemblages will likely reside along the sediments. This will be a result of increased temperature and progression to dormant/resting stages. The toxic effect of metals and other species will be reduced due to a reduction in metabolic activity and utilization of stored lipids.

Benthos

Benthic invertebrates will be subjected to toxicants in the sediment, porewater, and interstitial spaces

Benthic invertebrate production is, in part, a result of *in-situ* primary and secondary production. To compensate for lower production, many benthic invertebrates lower metabolism (Ward 1992). Benthic metabolism and productivity is not as reduced as water column species because they are able to feed on resting/dormant phytoplankton and zooplankton. Therefore, they may be more susceptible to the effects of toxicants in the sediment, porewater, and interstitial spaces.

Productivity and biomass for the benthic community is expected to be slightly reduced in the winter

Essentially, the benthic community in Snap and other regional lakes is expected to be slightly reduced both in terms of productivity and biomass during the winter. Where temperature effects are substantial, benthic organisms will migrate to warmer water. Important species in Snap Lake include the Dipterans, primarily Chironomidae. Members of the aquatic Diptera (Chironimidae) are able to survive freezing and live in deep water sediments (Chebucto 2001). Chironomidae are extremely adaptable and contribute a progressively greater proportion of the diversity and abundance of lake benthos from the sub-arctic to the high Arctic (Ward 1992).

9.5.2 Impact Assessment

9.5.2.1 Introduction

Traditional knowledge emphasized the importance of fish as an Aboriginal food

During community consultations and during the work with the Lutsel K'e Elders, it was made clear that fish is an important staple of people's diet.

Although people from Lutsel K'e do not fish much in the Snap Lake area, the waters of the Lockhart River provide a good portion of their fish (Lutsel K'e Dene First Nation 2001).

> *This water (in the Na Yaghe Kue region) (Snap Lake area) is important, because of all the fish and drinking water. Everywhere you go through all the lakes (in the Lockhart River watershed) there are lots of fish – whitefish, grayling, loche, pike and lake trout* (JM 10 06 01) (Lutsel K'e Dene First Nation 2001).

Blasting was identified as an issue

In addition to their concern on protecting fish as a food source, the North Slave Métis are also concerned that fish could be affected from blasting activities.

Key questions for assessing impacts on water quality were developed

To address the concerns raised by traditional knowledge and to meet the EA Terms of Reference (Table 9.1-1 in Section 9.1), the issues were consolidated into key questions. Key questions for assessing the impacts of the Snap Lake Diamond Project aquatic organisms and habitat issues include the following:

Key Question F-1: What impacts will the Snap Lake Diamond Project have on the quality and quantity of non-fish aquatic organisms?

Key Question F-2: What impacts will the Snap Lake Diamond Project have on fish habitat?

Key Question F-3: What impacts will the Snap Lake Diamond Project have on acute or chronic effects on fish health?

Key Question F-4: What impacts will the Snap Lake Diamond Project have on fish abundance?

The assessment method consists of a series of six steps

The key questions are addressed by the following methodology:

- presentation of potential linkages between project activities, non-fish aquatic organisms, and fish and fish habitat;
- evaluation of the validity of each pathway linking activities to potential effects on non-fish aquatic organisms, and fish and fish habitat;
- descriptions of the mitigation measures that will be implemented to minimize potential impacts on non-fish aquatic organisms, and fish and fish habitat;

- descriptions of the non-fish aquatic organisms, and fish and fish habitat impact analysis methods and results;
- classification of the residual impacts; and,
- descriptions of the recommended monitoring programs.

Pathways linking project activities and the aquatic ecosystem are presented in Figure 9.5-10

Potential linkages between the Snap Lake Diamond Project activities, non-fish aquatic organisms, and fish and fish habitat are illustrated in Figure 9.5-10. Potential pathways linking project activities to each key question will be evaluated. Impacts will be quantified for key questions with one or more valid pathways.

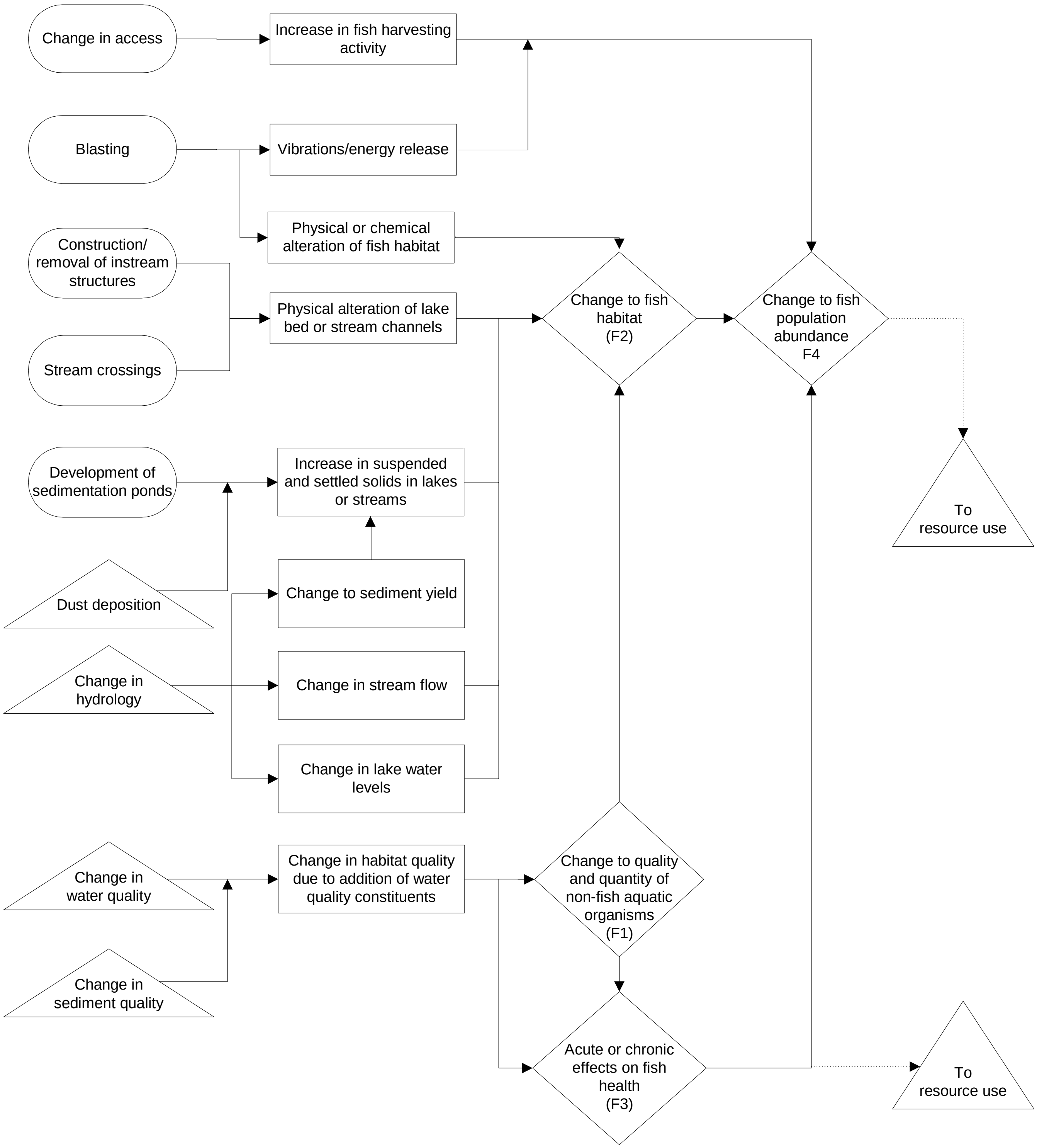
Potential effects to both non-fish aquatic organisms and fish were evaluated with respect to physical and chemical changes resulting from project activities

Potential effects on non-fish aquatic organisms (Key Question F-1) may occur through the alteration of water and sediment or physical changes in habitat. Physical changes to fish habitat (Key Question F-2) may occur as a result of several activities including in-stream (or in-lake) construction, blasting, stream crossings, and changes in hydrology. Fish habitat refers to spawning, nursery, rearing, food supply, overwintering, and migration areas required by fish. Indirect effects on fish habitat may occur because of changes in the abundance or quality of non-fish aquatic organisms that are consumed by fish. Acute and chronic effects on fish health (Key Question F-3) may arise from changes in water quality related to water use and water releases. Effects on the abundance of fish populations (Key Question F-4) may occur as a result of changes to fish habitat and changes to fish health, as well as the direct affects of blasting or increased fish harvesting.

Baseline information and established thresholds were used in the evaluation of effect levels

Impacts on both fish and non-fish aquatic organisms for the Snap Lake Diamond Project were assessed in relation to baseline conditions established during sampling in 1999-2001 and to available water and sediment quality criteria and site specific benchmarks for the chemicals of concern (see Section 9.4). Impacts from the uptake of chemicals of concern into aquatic organisms were assessed by evaluating the potential for increased concentrations of these chemicals in fish and then assessing whether these increased concentrations would affect the health of the fish. A similar approach of comparing project activities to established guidelines was used for evaluating the effect of blasting to the potential for direct fish mortality.

Figure 9.5-10 Aquatic Organisms and Habitat Linkage Diagram



The evaluation of changes to fish habitat were completed using a modified habitat evaluation procedure

Impacts on fish habitat, in areas where a potential loss of habitat may occur, were evaluated through a modified Habitat Evaluation Procedure (HEP). This approach calculates the quality and quantity of the habitat being altered, lost, or created during any phase of the project. HEP analysis combines habitat quality, defined as a Habitat Suitability Index (HSI) for each species and life stage, with habitat quantity to calculate Habitat Units (HUs) (Appendix IX.5-12). HUs provide a measure that accounts for both the quantity and quality of habitat available for a given species and life stage.

Habitat units under baseline conditions were compared with habitat units during each project phase

Comparing the numbers of HUs available under baseline conditions to those available during construction, operations, and post-closure, allows the quantification of the overall number of HUs altered, lost, or created by the proposed project, including mitigative measures. This enables an evaluation of the net affect of the proposed project on fish habitat.

Sediment deposition evaluated using established guidelines

In addition, effects to fish habitat through sediment deposition were evaluated from established thresholds for sedimentation of fish habitat as well as general ecological principles related to this type of disturbance. The threshold implemented is based on a study that found that an accumulation of 1 mm of sediment was sufficient to cause decreased fish egg survival (Fudge and Bodaly 1984). Sedimentation in Snap Lake was calculated based on the annual depth of sediment accumulation from the dust deposition modelling.

The overall potential for a change to fish populations was evaluated through a review of both direct and indirect effects

The overall response of the fish population to potential direct effects, and to any indirect combined effects on fish health and fish habitat, were evaluated through comparison to general ecological principles. The estimation of impact magnitude at the fish population level incorporates fish mortality. Project activities that lead to an increase in mortality rates above levels the population could sustain would result in an impact. This evaluation was done with a weight of evidence approach.

9.5.2.2 Key Question F-1: What Impacts Will the Snap Lake Diamond Project Have on the Quality and Quantity of Non-fish Aquatic Organisms?

Changes to water and sediment chemistry have the potential to affect the quality and quantity of aquatic organisms

Aquatic organisms are fundamentally important for both their intrinsic value and as a fish food. Any alteration of their abundance or community structure may affect fish heath and abundance. Two linkages and numerous pathways are evaluated in this section.

9.5.2.2.1 Linkage Analysis

Changes in water and sediment quality were included in the linkage analysis

The following potential linkages between the Snap Lake Diamond Project and non-fish aquatic organisms were analyzed:

- linkage between alterations to water chemistry and effects to quality and quantity of non-fish aquatic organisms; and,
- linkage between alterations to sediment chemistry and effects to the quality and quantity of non-fish aquatic organisms.

Quality and quantity are defined

Alterations in community quantity include shifts in population abundance or biomass. Alterations in quality include shifts in community composition (*i.e.*, species composition) and toxicity.

Alterations to Water Chemistry

Alterations in water chemistry may affect non-fish aquatic organisms

The proposed Snap Lake Diamond Project will release chemical and biological elements to Snap Lake and other local study area (LSA) lakes that may alter water chemistry**.** Potential chemical sources were evaluated in Section 9.4.2 (Water Quality Impact Assessment). Potential pathways of water/sediment impacts include the following:

Pathways during operation:

- combined discharge from the water and sewage treatment plants;
- site runoff;
- seepage from the north pile and water management pond; and,
- atmospheric deposition of dust and associated chemical constituents.

Pathways during post-closure:

- site runoff;
- discharge from the water management pond;
- runoff from the north pile to the north arm of Snap Lake;
- seepage from the north pile into the north arm of Snap Lake; and
- atmospheric deposition of dust and associated chemical constituents.

Water quality constituents that may have an impact are listed

Based on Section 9.4.2, the following water quality constituents were evaluated as having a potential affect on water quality in the LSA:

- Hexavalent chromium in Snap Lake, evaluated as having a low impact magnitude and low environmental consequence.
- Total dissolved solids (TDS) in Snap Lake. TDS concentrations are predicted to increase from the baseline concentration of approximately 15 mg/L to a maximum average concentration of about 330 mg/L.
- Trivalent chromium in the north lake, NL5, and NL6 water column, evaluated as having a moderate impact magnitude and low environmental consequence.

Alterations to Sediment Chemistry

Alterations in sediment chemistry may affect community composition and abundance of non-fish aquatic organisms

Alteration of sediment chemistry has the potential to affect non-fish aquatic organisms through changes in quantity and quality of the organisms. These effects are important to aquatic ecosystem structure because of potential alterations to the availability and toxicity of preferred food species for fish. Sediment and porewater chemistries are directly affected by groundwater flux and chemical modifications to surface water. The linkage between changes to sediment chemistry and the quality and quantity of non-fish aquatic organisms is a valid pathway. Based on the assessment conducted in Section 9.4.2, several water quality constituents were evaluated as having a potential affect on porewater and sediment quality in the LSA. These include:

- Snap Lake sediment chromium and TDS;
- north lake and northeast lake trivalent chromium, evaluated as having a moderate impact magnitude and moderate environmental consequence;
- north lake and northeast lake aluminum, copper, and molybdenum, evaluated as having a low impact magnitude and low environmental consequence;
- north lake and northeast lake hexavalent chromium, evaluated as having a low impact magnitude and low environmental consequence; and,
- north lake and northeast lake nitrate and pH concentrations.

9.5.2.2.2 Mitigation

Collection, treatment, and outflow design of mine and wastewater discharge are key mitigation measures

The treatment and discharge of water from the site is evaluated and discussed in Water Quality (Section 9.4). The magnitude of chemical release will be mitigated through collection and treatment of mine water, surface runoff, and domestic effluent. Water from the water treatment plant and the sewage treatment plant will be combined into a single discharge pipe and passed through a diffusion structure. Diffusing the discharge will limit acute localized effects within Snap Lake.

Chemical pathways eliminated in Section 9.4

Seepage from the north pile and waste management pond is limited in quantity and constitutes a minor fraction of chemical loading. In addition, seepage constituents are efficiently filtered by wetlands biota and do not drain into sensitive aquatic habitat. Detailed seepage analysis in relation to water quality may be found in the Water Quality section of this report (Section 9.4).

Air emission containment and mitigation measures limit airborne deposition

Atmospheric deposition represents a potential loading source of metals and nutrients to Snap Lake via dust. Elevated dust and industrial emissions are a result of site operations including road use, waste incineration, power generation, quarrying, and underground mining. Mitigation of dust emissions is achieved through underground mining in a wet environment. Conveyer belts will be used to transport ore from underground crushers to the processing plant, eliminating the need for trucks to haul the ore to the surface and further reducing dust generation. Additionally, wet processing of ore will minimize dust formation. Other dust suppression measures include watering of the airstrip/roads and minimizing the exposed areas on the north pile. Acid deposition will be limited by maximizing energy efficiency and minimizing fuel use.

9.5.2.2.3 Impact Analysis

The water quality assessment formed the basis for the assessment of impacts on non-fish aquatic organisms

The water quality assessment provided an initial screening of predicted maximum chemical concentrations compared to CCME guidelines, US EPA criteria and site-specific benchmarks (depending upon the chemical). This screening, in combination with the other residual impact criteria produced overall impact ratings for each potential chemical of concern. Chemicals carried forward from the water quality assessment to the assessment of impacts on non-fish aquatic organisms were those that exceeded the "negligible" residual impact category. The following chemicals were carried forward to the non-fish aquatic organisms assessment:

Construction and operations:

- hexavalent chromium in the water column and sediment of Snap Lake; and,
- total dissolved solids in the water column of Snap Lake.

Post closure:

- trivalent chromium in the water column of the north lake, NL5, NL6, and in the porewater/sediment of the north lake and northeast lake;
- the following chemicals in the porewater/sediment of the north and northeast lakes:
  - hexavalent chromium;
  - aluminum;
  - copper;
  - molybdenum;
  - nitrate; and,
  - pH.

The magnitude criteria used in the assessment of impacts included consideration of exceedance of chronic effect values for each major group of organisms, as well as duration and extent of exposure

Magnitude criteria used in the assessment of impacts on non-fish aquatic organisms went beyond the community-level benchmarks used in the water quality assessment (HC5, HC10 *etc*.) to ensure that there was adequate consideration of the potential for impacts on sensitive, potentially keystone species within each major group (phytoplankton, zooplankton, benthos). Chronic effect values for each of the three major groups were identified from the literature. In addition, the estimate of impact magnitude had to account for the fact that impacts would be a function of dose, not simply concentration. Dose is determined by chemical concentration, duration of exposure, and spatial extent of changes in chemical concentration. Therefore, overall impact magnitude was determined by the particular combination of concentration relative to chronic effect values, spatial extent within one lake, and duration of exposure ($\leq$1 year). These combinations were as follows:

Negligible magnitude: maximum predicted concentrations less than chronic effect value (or LOEC if chronic effect value not available) in less than 1% of the lake; seasonal changes in water quality only;

Low magnitude: maximum predicted concentrations that exceed the chronic effect value (or LOEC) in less than 10% of the lake; seasonal changes in water quality only;

Moderate magnitude: maximum predicted concentrations that exceed one chronic effect value (or LOEC) in less than 20% of the lake; seasonal changes in water quality; and,

High magnitude: maximum predicted concentrations that exceed the chronic effect value (or LOEC) in 20% or more than 20% of the lake; year-round effects on water quality.

Comparing predicted concentrations with published chronic effect values was the first step in assessing magnitude of impacts from exposure to metals

The first step in assessing the magnitude of potential impacts of metals on non-fish aquatic organisms was to compare predicted concentrations with published chronic effect values for relevant species. The "chronic effect value" is the geometric mean of the Lowest Observed Effect Concentration (LOEC) and the No Observed Effect Concentration (NOEC) (Suter and Tsao 1996). The chronic effect value data were evaluated for relevance to Snap Lake and north lake species using the general rules for using a subset of toxicological data (MacDonald *et al.* draft) (see Appendix IX.8 for details). If there were insufficient data to calculate chronic effect values, estimated chronic effects thresholds were derived using results for the most sensitive species reported in the literature. Data for irrelevant species were not included.

The potential magnitude of impacts of TDS on non-fish aquatic organisms was assessed by comparing predicted concentrations with published LOECs from laboratory studies and with observational data from the field

The potential impacts of TDS on non-fish aquatic organisms were assessed by comparing predicted concentrations with published LOECs from laboratory studies and with observational data from the field. Chronic effect values could not be derived for TDS because of the lack of data. There are almost no laboratory studies of TDS; rather, laboratory tests are usually done on individual ions such as chloride. Therefore, LOECs for the ions contributing the largest portion to TDS were examined. The dominant ions are calcium and chloride. Observational data from the field usually refer to TDS; however the constituents of the TDS vary from sodium chloride to sodium/magnesium to sodium bicarbonate lakes. Observational data of the effects of road salting were also useful; however, these data focussed on chloride effects rather than the cations associated with chloride.

If predicted concentrations exceeded chronic effect values, then the spatial and temporal extent of exceedances was examined

The magnitude of impacts depends not only on exceedance of chronic effect values, but also on the area or volume of the lake affected by these exceedances, and the length of time over which the exceedances occur. The area or volume of the lake affected is important because effects on a relatively small volume could be compensated by production of these organisms (which are primarily r-strategists with high reproductive rates) in the remainder of the lake. Furthermore, the predicted changes in water quality in Snap Lake are confined to deep basin areas (below 8 m depth) in winter. These deep basin areas are unlikely to represent critical fish feeding habitat. The length of time that metals or TDS are expected to be elevated is also important to the estimation of magnitude. Elevated metal or TDS concentrations over one season would be expected to have a lower magnitude of impact than elevated concentrations persisting year-round.

The season when increased concentrations occur is also important; increased concentrations during the winter when most organisms are less active and are not reproducing may have lower impacts than increased concentrations during the spring-to-fall period.

Other impact assessment criteria were identical to those used in the water quality assessment

The other impact assessment criteria were identical to those used in the water quality assessment as well as the assessment of fish habitat and fish health (Table 9.1-3). Direction was either neutral (no change in non-fish aquatic organism abundance or diversity) or negative (a decrease in abundance or diversity). Geographic extent was either local (restricted to Snap Lake or the lakes north of Snap Lake) or regional (the RSA). Duration was either short-term (pre-construction and construction), medium-term (the 26 years of operation) or long-term (26+ years). For post closure effects on the lakes north of Snap Lake, long-term is expected to be much longer than 26+ years, although the length of time is uncertain. Reversibility was short term (reversible within 30 years), long-term (reversible in greater than 30 years) or irreversible. Frequency was low (occurs once), medium (occurs intermittently) of high (occurs continuously).

The overall assessment of residual impacts was conducted after the impact magnitude for each of the chemicals of concern was rated for each major group

The overall assessment of residual impacts on non-fish aquatic organisms took the results of the assessment of impact magnitude for each of the chemicals of concern and then carried the evaluation through the remainder of the residual impact criteria. For example, the magnitude of impacts from chromium concentrations in Snap Lake was combined with the rating for geographic extent, duration, reversibility, and frequency to derive the overall rating for environmental consequence. These overall assessments are found at the end of this section. The bulk of this section deals with impact magnitude because this is the criterion that drives the overall environmental consequences.

Each of the three major groups of non-fish aquatic organisms was evaluated separately for each chemical of concern

Each of the three major groups of non-fish aquatic organisms (phytoplankton, zooplankton, benthos) was evaluated separately for each chemical of concern using the above step-by-step process. This process was followed for Snap Lake Construction and Operations and for Post Closure. The process can be illustrated as follows:

- Impact Analysis
  - Chemical of Concern
    - Phytoplankton
      - Comparison with chronic effect values
      - Spatial and temporal extent (if chronic effect values are exceeded)
      - Impact magnitude rating

- Zooplankton
  - As for phytoplankton
- Benthos
  - As for phytoplankton

- Residual Impact Classification
  - Chemical of Concern
    - phytoplankton
    - zooplankton
    - benthos

Snap Lake During Construction and Operations

Hexavalent Chromium

Direct exposure of chromium via water and sediment is the primary pathway of concern

Exposure to hexavalent chromium will occur primarily via the water column; however, sediment exposure may also occur. Some of the chromium present as fine particulate matter is expected to settle to the bottom, thus increasing sediment concentrations. To provide conservative estimates of impact magnitude, Snap Lake sediment and porewater concentrations are assumed to be similar to water column concentrations.

Water quality impact magnitude of hexavalent chromium concentrations in Snap Lake was rated as low

The maximum discharge hexavalent chromium concentration is predicted to be 7.51 µg/L. Predictive modelling estimates the maximum hexavalent chromium concentration 230 m from the discharge to be approximately 2.54 µg/L. Based on an exceedance of the HC_5 in less than 3% of the lake, and an exceedance of the HC_{10} in less than 1% of the lake, the impact magnitude to surface water quality was rated as low (Section 9.4.2).

Phytoplankton

Site specific water quality benchmarks were not created for phytoplankton species

United States Environmental Protection Agency (U.S. EPA) and Canadian Council of Ministers of the Environment (CCME) water quality guidelines were created primarily for Kingdom Animalia. Chromium toxicity to plants must be weighed against available, but limited, data. Available phytoplankton chromium data are reported as total chromium. Therefore, total chromium data were used for assessing the effects of both hexavalent and trivalent chromium.

Total chromium is not as toxic to plants as it is to animals

Trivalent and hexavalent chromium effects are manifested by decreased chlorophyll, carotenoid, protein content, oxygen evolution, and nitrate reductase activity (Stevenson *et al*. 1996). The lowest reported chronic effect threshold for algae reported in the literature is 220 µg/L for the dinoflagellate *Gymnodinium splendens* (Wilson and Freeburg 1980). In lieu of a chronic effect value, this LOEC of 220 µg/L was used to evaluate impact magnitude.

The impact magnitude to Snap Lake phytoplankton is negligible

The Snap Lake maximum total chromium concentration is predicted to be 7.5 µg/L in 1% of the lake and 2.5 µg/L in less than 3% of the lake. The Snap Lake chromium concentration is predicted to be well below the estimated toxicity threshold of 220 µg/L. The impact magnitude for Snap Lake phytoplankton exposed via water or sediments is, therefore, rated as negligible.

Zooplankton

The impact magnitude of hexavalent chromium to Snap Lake zooplankton is negligible

The maximum water column concentration of chromium 230 m is expected to be 2.5 µg/L. This concentration is less than the chronic effect values for the three most sensitive species reported in the literature. These three species are all Cladocerans: *Daphnia magna* (3.32 µg/L), *Daphnia pulex* (6.13 µg/L) and *Simocephalus vetulus* (6.13 µg/L). Based on this, the impact magnitude of hexavalent chromium on Snap Lake zooplankton exposed via water or sediments is negligible.

Benthos

The impact magnitude of hexavalent chromium to Snap Lake benthos is negligible

All benthic species have chronic values above the maximum discharge concentration of 7.51 µg/L (Appendix IX.8, Tables IX.8-5 and IX.8-6). Dominant taxa in Snap Lake include *Tanytarsus* and *Chironomus*, which have chronic concentrations of 9,679 µg/L and 10,304 µg/L, respectively. The most susceptible species in Snap Lake is *Hyalella azteca* (106.42 µg/L) and the most susceptible amphipod evaluated is *Gammarus pseudolimnaeus* (11.33 µg/L). All Snap Lake benthic species, and all species used to establish site-specific benchmarks, have chronic toxicity values above the discharge concentration. Therefore, the magnitude of effect of hexavalent chromium to Snap Lake benthos via water or sediment is negligible.

Total Dissolved Solids

There are no general or site-specific criteria for total dissolved solids

There is no aquatic biota guideline or site-specific criterion for total dissolved solids (TDS). TDS, an expression of salinity, is a measure of the sum of all dissolved ionic constituents in water. The average baseline concentration for TDS in Snap Lake is very low at 15 mg/L. The proposed Snap Lake Diamond Mine is expected to raise the TDS concentration to a whole lake maximum of 330 mg/L (year twenty) in 10-20% of the lake area. The ions responsible for this increase will be calcium and chloride. The primary toxicological concern of TDS is an increase in osmotic stress on aquatic biota.

Phytoplankton

A threshold for defining a saline lake is 3000 mg/L TDS

There are no TDS water quality criteria, but an accepted threshold for classification of a saline lake is 3,000 mg/L (Hammer *et al* 1975, Timms *et al* 1986, Mandaville 2002). At TDS concentrations greater than 3,000 mg/L, freshwater biota begin to disappear (Hammer *et al*. 1975).

The impact magnitude of TDS to Snap Lake phytoplankton is negligible

Predicted TDS concentrations in Snap Lake are less than upper limits for relevant phytoplankton species. Wilson *et al*. (1994, 1996) determined a salinity optimum, lower and upper limits for selected phytoplankton species (Table 9.5-18). Since the predicted maximum TDS concentration is much less than the upper limits and slightly higher or similar to salinity optima for these species, the impact magnitude to Snap Lake phytoplankton is predicted to be negligible.

Table 9.5-18 Salinity Optima for Phytoplankton Species

Taxon	Salinity Optimum (mg/L)	Lower Limit (mg/L)	Upper limit (mg/L)
Tabellaria flocculosa	180	0	10,090
Cyclotella bodanica	240	30	1,820
Fragilaria tenera	320	10	807

Adapted from Wilson *et al*. (1994, 1996).
mg/L = milligrams per litre.

Zooplankton

Effects on zooplankton do not occur until concentrations exceed 1200 mg/L TDS (as NaCl)

Cowgill and Milazzo (1990) determined the toxicity of salinity to various cladocerans (water fleas). The sodium chloride (NaCl) LC_{50}'s for *Daphnia magna* (6,034 mg/L) and *Ceriodaphnia dubia* (2,019 mg/L) were very high. Hammer *et al*. (1975) determined that the rotifer *Keratella quadrata* (present in Snap Lake) thrives in highly saline environments of 0.035-2.761 eq/L (2,000-3,000 grams per litre [g/L]). Cowgill and Milazzo (1990) determined that a conservative "no observable effect level" to avoid reproductive limitation would be ~1,200 mg/L NaCl. This is much higher than the predicted TDS concentration of 330 mg/L for Snap Lake; however the TDS in Snap Lake will primarily consist of calcium chloride, not sodium chloride.

Analysis of impacts of the individual ions is necessary to increase confidence in the assessment

Consideration of the individual ions contributing to the TDS increase in Snap Lake is required to increase the confidence in the assessment of impact magnitude on zooplankton. The TDS threshold for effects on zooplankton is far above predicated concentrations, leading to an initial impact magnitude rating of negligible. However, this is re-evaluated in the following section on the two primary ions contributing to TDS, calcium and chloride.

Benthos

Field data indicate that effects from salinity on benthos do not occur until concentrations far exceed predicted Snap Lake concentrations

Observational data from field studies indicate that effects from salinity do not occur until concentration are much higher than the predicted 330 mg/L for Snap Lake. Hynes (1990) describes no effects on the benthic invertebrate community of a lake in northern Saskatchewan receiving treated uranium mill effluent with elevated TDS. In this study, the increase in total dissolved solids from baseline conditions was from 76 mg/L to 2,700 mg/L. The major ions primarily responsible for this increase were calcium, sodium, chloride, and sulphate. There were no statistically significant decreases in abundance or species diversity. Species richness declined, with fewer oligochaetes, Hirudinea, and amphipods, but considerably more *Tanytarsus* (Chironomidae).

Predicted TDS concentrations do not approach thresholds for effect, but the ionic mixture in Snap Lake will differ from that in the literature

Predicted TDS concentrations do not approach thresholds for effect; however, most data are for ionic mixtures that are different than the mixture predicted for Snap Lake. The ions making up the TDS in laboratory and field studies are usually sodium chloride, sodium or magnesium sulphate or potassium chloride. Calcium chloride effects have rarely been studied. The predicted TDS concentrations in Snap Lake are due largely to calcium chloride. Therefore, the impact magnitude of effects on benthos is re-evaluated in the following section.

TDS: Ionic Constituents

The primary ions in freshwater lakes that make up the TDS measurement include:

1. cations: Ca^{2+}, Mg^{2+}, Na^{+}, K^{+}; and,
2. anions: HCO_3^{-}, CO_3^{2-}, SO_4^{2-} and Cl^{-}.

The ions contributing most of the increase in TDS in Snap Lake are calcium and chloride.

Concentrations of calcium are predicted to increase from 1.3 to 110 mg/L in 10-20% of Snap Lake and chloride concentrations from <0.2 to 140 mg/L in 10-20% of Snap Lake

The average baseline concentrations for calcium and chloride are very low at 1.34 mg/L and <0.2 mg/L, respectively. The proposed Snap Lake Diamond Project is expected to raise water column calcium concentrations to 80-90 mg/L and chloride concentrations to 120 mg/L within 1,000 m of the outlet. The maximum concentration of calcium is expected to be 110 mg/L in 10%-20% of the lake area. The 110 mg/L maximum concentration is a product of effluent concentration and wind dynamics in operation years 15-25. The maximum chloride concentration is expected to reach 140 mg/L in 10-20% of the lake area.

Chloride

Phytoplankton, Zooplankton, and Benthos

Chloride chronic effects begin at concentrations greater than 372 mg/L

The chloride ion is considered to have chronic effects on aquatic organisms above concentrations of 372.1 mg/L (U.S. EPA 1988). The water quality guideline for the protection of aquatic life is 230 mg/L. The Snap Lake peak concentration is 140 mg/L.

Calcium

Effects of increased chloride concentrations on non-fish aquatic organisms are expected to be negligible

Effects of increased chloride concentrations on phytoplankton, zooplankton, and benthic organisms are expected to be negligible. Maximum predicted chloride concentrations will not approach lowest observable effects concentrations (LOECs) reported in the literature, even for the most sensitive species.

The primary cation of forest/tundra lakes is often calcium

The primary calcium species found in freshwaters include a variety of calcium carbonates, bicarbonate complexes, hydroxides, and sulfates and to a lesser extent chlorides, bromides, and fluorides. The Snap Lake ion balance is similar to forest/tundra lakes in the forest/tundra ecoregion (Table 9.5-19) Calcium is the primary cationic constituent.

Table 9.5-19 Cation Comparison Data for Snap Lake and Other Forested Tundra Lakes

Summary Balance Comparison					
Forest Tundra Lakes			Snap Lake		
Cations	Average (mg/L)	Percent	Cations	Average (mg/L)	Percent
Ca2+	2.01	54%	Ca2+	1.34	49%
Na+	0.78	21%	Na+	0.57	21%
K+	0.59	16%	K+	0.44	16%
SiO_2	0.32	9%	SiO_2	0.4	14%
Cl-	0.69		Cl-	<0.2	

Adapted from Pienitz *et al*. 1997.
mg/L = milligrams per litre.

Calcium is an essential nutrient but can cause reproductive effects in zooplankton when present in excess

Calcium is an essential nutrient but in excess it is known to cause reproductive effects (Biesinger and Christensen 1972). In general, copepods and *Daphnia* are more susceptible to toxicological effects than other zooplankters, namely rotifers.

Phytoplankton

Phytoplankton appear to be tolerant to a wide range of calcium concentrations; therefore impact magnitude is negligible

Vyverman *et al*. (1996) found limited differences in phytoplankton assemblages attributable to calcium concentrations. Species richness was not affected until concentrations were very high. Diatom richness was not substantially affected in calcium ranges from 8-203 ueq/L (0.16-4.06 g/L). Of interest, Vyverman *et al.* (1996) concluded the mean calcium effect concentration for *Tabellaria flocculosa* to be 44.26 meq/L and the lower tolerance threshold to be 32.12 meq/L. These thresholds are far above predicted calcium concentrations; therefore, the magnitude of impact on phytoplankton is estimated to be negligible.

Zooplankton

Maximum calcium concentrations in Snap Lake will be approximately equivalent to the published Lowest Observed Effect Concentration for Cladocerans

Maximum predicted calcium concentrations of 110 mg/L will be approximately equivalent to the published LOEC of 116 mg/L for *Daphnia magna* (Biesinger and Christensen 1972). However, the maximum concentration would only occur in 10-20% of the lake area in the winter months. Furthermore, the average predicted calcium concentration (90 mg/L) would be well below the LOEC for *Daphnia magna*.

The magnitude of calcium effects on sensitive zooplankton species is expected to be low

Based on the above information, calcium effects on some zooplankton species are expected to be low. This conclusion is conservative, since maximum predicted concentrations are roughly equivalent, but do not exceed, the LOEC for sensitive cladoceran species. Therefore, based upon no exceedance of chronic effect values, the magnitude would be rated as negligible. However, the spatial extent of increased calcium concentrations would be from 10-20% of the lake area, albeit only in the winter months. In order to be rated negligible, spatial extent should be less than 1% of the lake. Therefore, the overall rating for zooplankton impact magnitude was determined to be low.

No chronic effects data were found in the literature; therefore impacts magnitude was based on Daphnia Magna data.

Benthos

There are no published studies of the effects of calcium on benthic invertebrates. Therefore, the impact magnitude was evaluated using the effect threshold from *Daphnia magna*. This is a conservative approach, since cladocerans are usually among the most sensitive invertebrate taxa.

The magnitude of impacts on benthic invertebrates is conservatively estimated as low

Calcium impacts on benthic invertebrates are conservatively estimated to be of low magnitude. Maximum calcium concentrations are roughly equivalent to, but do not exceed, the LOEC for *Daphnia magna*. Therefore, there is a small potential for impacts on benthic invertebrate species which are as sensitive as *D. magna*. The potential effects would occur in 10-20% of the lake area in the winter months. This temporal and spatial extent places the magnitude in the low category.

Post-Closure

Hexavalent Chromium

Predicted sediment/ porewater hexavalent chromium concentrations in the north and northeast lakes are 313 µg/L

The post-closure maximum concentration of hexavalent chromium in the porewater/shallow bottom layer under ice is 313 µg/L for both the north and northeast lakes (Section 9.4.2). The area affected is less than 10% but greater than 1% of the sediment area that receives groundwater affected by the underground mine workings. Effects of porewater hexavalent chromium are described, where appropriate, for phytoplankton, zooplankton and benthic organisms.

Phytoplankton

Effects of total chromium concentrations were estimated

As explained for Snap Lake, there are no chromium criteria for phytoplankton. Furthermore, toxicity data are for total chromium, not trivalent or hexavalent chromium. Therefore, this assessment uses an estimated chronic toxicity effect threshold of 220 µg/L for total chromium.

The impact magnitude to phytoplankton in the north and northeast lakes is moderate

The north and northeast lake total chromium sediment porewater concentrations are predicted to reach a maximum of 320.5 µg/L, which is above the predicted phytoplankton chronic toxicity threshold. Based on this conservative threshold, the primary effects will be a slight reduction in biomass in less than 10% of the lake area; however, these effects would be year round. This produces an impact magnitude rating of moderate.

Zooplankton

Impact magnitude of hexavalent chromium to north and northeast lake zooplankton is moderate

The maximum average porewater concentration of 313 µg/L in the north lake is higher than chronic values for the following cladocerans: *Daphnia magna* (3.32 µg/L), *Daphnia pulex* (6.13 µg/L), *Simocephalus vetulus* (6.13 µg/L) *Ceriodaphnia dubia* (10.00 µg/L), *Simocephalus serrulatus* (19.90 µg/L) and *Ceriodaphnia reticulata* (40.00 µg/L). Based on the regional baseline data, the dominant species expected to occur in the north lakes include the calanoid copepods *Leptodiaptomus sicilis* and *Leptodiaptomus minutus*. For the purposes of this assessment, it is assumed that copepods are as sensitive as cladocerans. The spatial extent of exceedances of the chronic effects value is <10% of the lake. The duration of effects would be year-round. The impact magnitude of hexavalent chromium in the north and northeast lakes is rated as moderate because although <10% of the lakes would be affected, these effects would be year-round. The effect expected is a reduction in zooplankton biomass. Bioaccumulation is not expected (Chapman *et al*. 1998). Preferential selection for rotifers (more tolerant) is expected over copepods and cladocera.

Benthos

The predicted magnitude of impacts on benthic organisms from elevated hexavalent chromium in porewater is moderate

Benthic organisms with chronic effect values below the predicted 313 µg/L on the Species Sensitivity Distribution for Hexavalent Chromium (Appendix IX.8, Table IX. 8-6) include *Gammarus pseudolimnaeus* (11.33 µg/L), *Hyallela azteca* (106.42 µg/L), *Gammarus fossarum* (190.28 µg/L), and *Heptagenia sulphurea* (225.43 µg/L). *Hyallela azteca* was noted in baseline data from five reference stations and had a sample population greater than one in only one station (five members in WQR 7-3 in the north lake). The dominant benthic organisms were midge larvae (Diptera; Chronomidae: Tantarsini, Chironomini). The chronic effect values for these tribes of midge larvae are 9,679 µg/L and 10,304 µg/L respectively. The predicted concentration of hexavalent chromium does not approach the chronic value for the dominant organisms in the north and northeast lakes. However, it does exceed the benchmark for effects on *Hyallela* (a species that was present but not abundant in baseline samples). The spatial extent of the exceedance of the chronic effect value for *Hyallela* would be less than 10% of either lake and effects are year-round. The resulting combination produces a moderate magnitude classification for the north and northeast lakes.

Trivalent Chromium

The north lake, NL5, and NL6 will have elevated trivalent chromium concentrations in the water column

The water column concentration of trivalent chromium in the north lake is predicted to be 12.3 µg/L. The concentration of trivalent chromium in NL5, which appears to receive all its flow from the north lake, will likely approach the concentration in the north lake. The concentrations in NL6 will also be similar, but may be slightly lower because NL6 receives some run-off from a separate sub-basin.

They will have elevated trivalent chromium in the sediment

The maximum porewater concentration of trivalent chromium in the north lake is predicted to be 12.5 µg/L. The maximum porewater concentration in the northeast lake is predicted to be 4.5 µg/L.

Phytoplankton

Effects of trivalent chromium were estimated using a total chromium toxicity threshold

As explained previously, there are insufficient data to derive a chronic toxicity threshold for trivalent chromium effects on phytoplankton. Therefore, the LOEC for total chromium of 220 µg/L is used to assess the magnitude of impact of increased trivalent chromium concentrations in both the water column and sediment porewater.

The impact magnitude for trivalent chromium effects on phytoplankton is negligible

The predicted maximum concentrations of trivalent chromium in the water column of the north lake, NL5, and NL6 and in the sediment porewater of the north and northeast lakes are well below the LOEC of 220 µg/L. Therefore, the impact magnitude of trivalent chromium on phytoplankton is rated as negligible.

Zooplankton

The chronic effect value for relevant zooplankton species is 40 ug/L

The chronic effect value for relevant zooplankton species is 40 µg/L, based upon toxicity tests using *Daphnia magna* (Chapman *et al*. Manuscript). A much higher chronic effect value of 225 µg/L has been reported for *Daphnia pulex* (Stackhouse and Benson 1989); however, the lower value for *D. magna* was selected as a conservative estimate.

The impact magnitude for trivalent chromium effects on zooplankton is negligible

The predicted maximum concentrations of trivalent chromium in the water column of the north lake, NL5, and NL6 and in the sediment porewater of the north and northeast lakes are well below the chronic effect value of 40 µg/L. Therefore, the impact magnitude of trivalent chromium on zooplankton is rated as negligible.

Benthos

The chronic effect value for effects on benthic invertebrates is 16 µg/L

The chronic effect value for relevant benthic invertebrate species is 16 µg/L, based upon toxicity tests using the mayfly *Ephemerella subverla* (Warnick and Bell 1969). Other relevant benthic chronic effect values include: 21 µg/L for *Tubifex tubifex* (Fargasova 1994), 24 µg/L for *Gammarus* (Rehwoldt et al. 1973) and 81 µg/L for the midge *Chironomus* (Rehwoldt *et al*. 1973).

The impact magnitude for trivalent chromium effects on zooplankton is negligible

The predicted maximum concentrations of trivalent chromium in the water column of the north lake, NL5 and NL6 and in the sediment porewater of the north and northeast lakes are below the chronic effect value of 16 µg/L. Therefore, the impact magnitude of trivalent chromium on benthic invertebrates is rated as negligible.

Copper

Porewater copper concentrations are carried forward from the water quality assessment

The maximum water column concentration for copper is predicted to be 1.1 µg/L in the north lake and 1.0 µg/L in the northeast lake. No effect is predicted for NL5 or NL6. The maximum porewater concentration is predicted to be 5.1 µg/L in the north lake and 5.1 µg/L in the northeast lake.

Phytoplankton

Copper is a well known phytoplankton algicide

Copper sulfate ($CuSO_4$) has traditionally been used as an algicide, although its use has been prohibited in several states of the U.S.A. The most toxic form of copper to algae is cupric ion, Cu^{2+} (Cook *et al*. 1993). The primary effects to algae include inhibition of photosynthesis, cell division, and nitrogen fixation (Stevenson *et al*. 1996).

The species of available copper varies according to pH

The pH in the north and northeast lake sediments is predicted to reach a maximum of 11.8. In low alkalinity waters, the primary species of copper above pH 10 include Tenorite ($CuO_{(s)}$) and $Cu(OH)_3^-$ (Morel and Hering 1993). There is essentially no Cu^{2+} available above pH 9 in low alkalinity waters and $1x10^{-7}$ available above pH 10.5 in high alkalinity water (Cooke 1993).

Diatoms and some green algae are more susceptible to copper toxicity

Ionic copper levels as low at 0.1 µg/L can kill some algae in water with low chelation potential (Horne and Goldman 1994). The most copper sensitive saltwater diatom, *Thalassiosire pseudonana*, experience growth inhibition at concentrations of copper as low as 5 µg/L (Erickson 1972). This finding is in-line with the most sensitive freshwater algae. According to U.S. EPA (1980), mixed culture algae exhibited reduced photosynthesis at 5 µg/l, *Chlorella* sp. showed reduced growth at 1 µg/L, and inhibited photosynthesis at 6.3 µg/L, and the freshwater diatom *Nitzschia palea* displayed complete growth inhibition at 5 µg/L. Therefore, the LOEC for algae chronic effects appears to be 1 µg/L, based on growth inhibition in *Chlorella* sp.

The impact magnitude of copper on north lake phytoplankton is low

Some sensitive algal species in the north and northeast lakes will be susceptible to copper toxicity. This is a conservative assessment based on the LOEC from the green algae *Chlorella* of 1 µg/L. The predicted water column concentrations in the north and northeast lakes will be approximately equivalent to the LOEC. Predicted porewater concentrations of about 5 µg/L will exceed this LOEC and approach the LOEC for effects on susceptible diatom species. Given that the toxicity would be present in greater than 1% but less than 10% of the sediment area, the impact magnitude is low. Notwithstanding that the duration of exposure is year-round, the impact magnitude was rated as low because the pH-adjusted copper ion concentration will be greatly reduced.

Zooplankton and Benthos

Chronic toxicity values for zooplankton and benthos

The chronic toxicity values of relevant zooplankton and benthic species range from 1.6 to 4.5 µg/L corrected for a hardness of 20 mg/L (Table 9.5-20).

Table 9.5-20 Predicted Chronic Sediment Toxicity Values for Copper

Species	Common Name	Copper Chronic Value at 20 mg/L Hardness (µg/L)
Ceriodaphnia dubia	Cladoceran	1.6
Daphnia pulicaria	Cladoceran	1.6
Ceriodaphnia reticulata	Cladoceran	1.7
Gammarus pseudolimnaeus	Amphipod	3.0
Moina dubia	Cladoceran	3.3
Daphnia magna	Cladoceran	3.8
Gammarus pulex	Amphipod	4.0
Daphnia ambigua	Cladoceran	4.3
Daphnia pulex	Cladoceran	4.4
Physa heterostropha	Snail	4.5

[1] Value derived from data in Appendix IX-8.

The copper chronic effect level for zooplankton is 1.6 µg/L

The lowest chronic effect value for *Daphnia* species is 1.6 µg/L (Table 9.5-20). *Daphnia longiremis* is present in Snap Lake and is expected to be present in the north and northeast lakes. Therefore, chronic effect levels based on *Daphnia* data are relevant.

The impact magnitude for copper effects on zooplankton is low

Predicted porewater concentrations of copper in the north and northeast lakes exceed the lowest chronic effect level. However, at the predicted pH level of 11.8, copper will not be present as the cupric ion. The cupric ion is the most toxic form of copper (Cook *et al*. 1993). Therefore, actual toxicity from the copper species present at pH 11.8 may be much lower than what is predicted based on toxicity values derived from tests where cupric ion would be the dominant copper species. The spatial extent of elevated copper concentrations in porewater is less than 10% of the north and northeast lakes. The duration of exposure will be year-round. On balance, despite the year-round exposure, the impact magnitude is rated as low because of the high likelihood that copper species present in the porewater will be less toxic than would be predicted under lower pH conditions. Little to no bioaccumulation of copper is expected because of the high pH.

The copper chronic effect value for benthos is 3.0 µg/L

The lowest chronic effect value for relevant benthic invertebrate species is 3.0 µg/L for *Gammarus pseudolimnaeus* (Table 9.5-20). *Gammarus* spp. were not found in the north lake during baseline sampling; however this chronic effect value is assumed to be relevant to *Hyalella* spp., which were present. The snail, *Physa* was also present in the north lake. The chronic effect value for *Physa heterostropha* is 4.5 µg/L.

The impact magnitude for copper effects on benthos is low

Predicted copper concentrations in porewater in the north and northeast lakes (5 ugh/L) will exceed the chronic effect level. However, as explained for zooplankton, it is highly unlikely that the copper will be present as the cupric ion at a pH of 11.8. Therefore, it is probable that the actual toxicity to benthic invertebrates will be lower than predicted using the chronic effect level. The spatial extent of elevated copper in the two lakes is less than 10%. Exposure will be year-round. On balance, despite the year-round exposure, the impact magnitude is rated as low because of the high likelihood that copper species present in the porewater will be less toxic than would be predicted under lower pH conditions. Little to no bioaccumulation of copper is expected because of the high pH.

Molybdenum

Porewater concentration exceeds the guideline

The predicted porewater concentration in the north and northeast lakes is 81.1 µg/L. This is above the CCME guideline at 73 µg/L. There are insufficient data to derive chronic effect values or site-specific benchmarks for molybdenum.

Molybdenum is a low toxicity essential nutrient

Molybdenum is a biologically essential micronutrient that is active in oxidation-reduction enzyme systems (CCME 1999, with 2000 updates). It is also a required element for nitrogen fixation. It is a low toxicity element that does not bioaccumulate in animal tissue. It does, on the other hand, accumulate in plant tissue. According to BC Environment (1995), molybdenum may become toxic to algal species above 54 mg/L. *Daphnia* have been reported to tolerate molybdenum concentrations in excess of 1,000 mg/L without perceptible injury (BC Environment 1995). The LC_{50} value determined for *Daphnia magna* at 48 hr was 2,847.5 mg/L (Diamantino *et al*. 2000), which is far above the predicted molybdenum concentrations in porewater of the north lakes. BC Environment (1995) determined "the proposed criteria for total molybdenum to protect sensitive aquatic life are an average value over thirty days not to exceed 1 mg/L, and a maximum value not to exceed 2 mg/L."

Phytoplankton, Zooplankton, and Benthos

The impact magnitude of molybdenum to phytoplankton, zooplankton and benthos is negligible

The molybdenum concentration in the north and northeast lakes sediments is slightly over the CCME guideline for aquatic life in less than 10% of the lake area. Despite the slight exceedance of the CCME guideline, the impact magnitude for effects of molybdenum on non-fish aquatic organisms is rated negligible for zooplankton and benthos because predicted concentrations will be far below published toxicity thresholds for relevant, sensitive species. However, the impact magnitude for phytoplankton is rated as low because predicted porewater concentrations exceed the BC Environment (1995) threshold for effects on algae. These exceedances would occur in less then 10% of the lake area, and would be year-round.

pH

Sediment pH is above normal range (6.3-9.6)

The sediment pH in the north and northeast lakes is expected to rise to 11.8. No pH effect is predicted in NL5 or NL6. This value is outside of normal pH range for North American lakes, (6.3 to 9.6). Pienitz (1997) *et al*. found the average pH (8.06) and pH range (7.5-8.8) in the Northwest Territories Forest-Tundra ecoregion to be slightly alkaline. The average baseline pH in the north lake is 6.6. The average pH (7.6) and pH range (6.3-8.6) of the Northwest Territories Arctic region is more typical of North American lakes. No site-specific benchmarks have been developed for pH but the CCME general guideline for aquatic life is 6.5-9.

Alkaline conditions will affect inorganic carbon constituents

A primary effect of pH will be on the forms of inorganic carbon at the sediment water interface. At a pH of 11.8 there is almost no carbon dioxide (CO_2), approximately 20% HCO_3^- and 80% CO_3^{2-} (Wetzel 2001). At this pH, the carbonate constituent is sufficient to form marl ($CaCO_{3(s)}$) in the presence of abundant calcium. The calcium content of the reference lake is low, but calcium scavenging by inorganic carbon will be expected along the sediment–water interface. In the absence of air equilibration, high pH will reduce the CO_2 available to algae to very low concentrations (Beardall *et al*. 1998). Ice-over in the north lake will prevent air equilibration.

Blue-green algae are favoured in low CO_2 concentrations

Cyanobacteria (blue-green algae) possess an environmental adaptation for survival at low CO_2 concentrations. The adaptation is known as CO_2 concentrating mechanism (CCM) and it functions to actively transport and accumulate inorganic carbon (HCO_3^- and CO_2; Ci) within the cell and then uses this Ci pool to provide elevated CO_2 concentrations around the primary CO_2-fixing enzyme, ribulose bisphosphate carboxylase-oxygenase (Rubisco) (Price *et al*. 1998).

North lake sediment pH levels are above most species ranges except for select bacteria

Natural alkaline lakes have pH ranges of 10-11 (Horne and Goldman 1994). Organisms which have adapted to alkaline environments are divided into alkali-tolerant organisms which grow best at pH 7.0-9.0 and cannot grow above pH 9.5 and alkalophiles which grow best between pH 10.0 and 12.0 (Ford 1993). The primary aquatic species that would survive in northern lake pH environments include *Bacillus aclalophilus* and *Actinomycete* sp. (pH 8.5-11.6 and 8.0-11.5, respectively) (Krulwich and Guffanti 1989). The algae *Chlorella pyrenoidosum* is very adaptable and will survive at pH ranges between 2.0 and 10.0 (Ford 1993).

Some species of algae with low pH preferences are present in lakes in the region

Chlorella sp. are not present in the regional lakes sampled but several diatom species with relatively high pH optima are. The most dominant are *Tabellaria fenestrata* with a relatively low pH optimum of 7.5 (dominant in Snap Lake) and *Navicula halophilia* with a pH optimum of 8.4 (Dixit *et al*. 1999).

Sedimentary phosphorus release rates rise at high pH values but may be controlled by calcium interactions

The influence of pH on sediment phosphorus release has been shown to be potentially significant (Wetzel 1983). Within the typical range of pH values for lakes, an increase in pH reduces the phosphorus sorption capacity of ferric iron (hydro)oxides, thereby increasing the potential for sediment phosphorus release (Penn *et al*. 2000). Phosphorus release may be controlled by interactions with calcium compounds which are also pH sensitive (Morel and Hering 1993). Increased sorption of phosphorus to $CaCO_3$, co-precipitation of phosphorus with $CaCO_3$ and hydroxyapatite ($Ca_5(PO_4)_3OH$) formation are all favoured at increased pH values (Penn *et al*. 2000). The calcium-phosphorus mechanisms may result in reduced rates of sediment phosphorus release at increased pH values (Penn *et al*. 2000). Any rise in phosphorus concentration may increase productivity (addressed below with nitrate).

High pH increases ammonia toxicity but reduces some metal mobility

An increase in pH may cause heightened ammonia concentrations (U.S. EPA 1986). Above a pH of 9, un-ionized ammonia (NH_3) is the predominant species (Morgan and Stumm 1981), which is very toxic to organisms (NRC 1997). However, high pH also reduces the toxicity of certain metals by binding them further. Higher pH is known to decrease cation mobility and increase anion mobility. The mobility of metals as hydrated ionic salts is dependent first upon which metallic element is participating as the positively charged ion and secondly, which anion makes up the negatively charged component of the salt (USGS 2002).

Phytoplankton, Zooplankton, and Benthos

The impact magnitude of pH on phytoplankton, zooplankton and benthos is moderate

Due to the general lack of toxicity data relating directly to sediment pH effects on aquatic biota, a plausible argument for impact magnitude must be developed. The north and northeast lake sediments will receive pH-affected waters in excess of the tolerance range of most phytoplankton, zooplankton and benthic species. A conservative impact magnitude rating of moderate is determined for pH effects for the north and northeast lakes sediments because of the potential for acutely lethal conditions caused by high pH. The elevated pH will occur in less than 10% of the aquatic sediments, year-round. Therefore, despite all the complex interactions between pH, carbon, phosphorus, nitrogen, and metals outlined above, the impacts from pH will be dominated by direct toxicity of the pH.

Nitrate

Nitrate concentrations will be much higher than baseline

The north and northeast lake sediment nitrate concentrations are expected to rise from <0.008 mg/L at baseline to 42.7 mg/L in less than 10% of the lake substrate. There are no current aquatic life guidelines for nitrate.

Phytoplankton

The north lake is N:P co-limited

Concentrations of nitrate-nitrogen range from undetectable levels to nearly 10 mg/L in unpolluted fresh waters but are highly variable seasonally and spatially (Wetzel 2001). The assimilation of nitrate and its reduction by green plants, in the presence of sufficient molybdenum, is the dominant process in the trophogenic zone of lakes. As seen previously, the molybdenum concentrations will not be limiting in this system. The north lake is co-limited as indicated by the baseline N:P stoichiometric ratio of 14 (Wetzel 1983).

Increased nitrate will favour Diatoms

An increase in nitrate will drive lake sediments to a more phosphorus-limited system. Diatoms, such as *Asterionella formosa* and *Cyclotella meneghiniana*, display luxury uptake of phosphorus (Horne and Goldman 1994). The luxury uptake coefficient (ratio of the cell quota of a nutrient when it is limiting to when it is abundant (Droop 1974)), for *Asterionella formosa* is 82 while the luxury uptake coefficient for *Cyclotella meneghiniana* is 6.6 (Tilman and Kilham 1976; Brown and Button 1979, respectively). Additionally, the minimum phosphorus requirement per unit cell volume increases along a species gradient: *Asterionella* (<0.2), *Fragiliaria* (0.2-0.35), *Tabellaria* (0.45-0.6), *Scenedesmus* (>0.5), *Oscillatoria* (>0.5) and *Microcystis* (>0.5) (Vollenweider 1968, Wetzel 2001).

The impact of nitrate on north and northeast lakes phytoplankton will be positive in direction and is predicted to be of low magnitude

Asterionalla formosa is one of the dominant species found regionally as would be expected to dominate in the north and northeast lakes. Other important algal species include *Fragilaria* and *Tabellaria*. A combination of increased soluble nitrogen in the presence of algae capable of maximizing low phosphorus concentrations will lead to an increase in production. *Asterionella formosa* will likely out-compete other algae and dominate the system. *Fragiliaria* and *Tabellaria* will likely grow in biomass and proportion. Production will potentially be increased by up to 30% and diatoms will dominate the algal community. The impact of an increase in nitrate to the north lake sediments is positive in direction. The impact magnitude is expected to be low based on an increase in diatoms over less than 10% of the lake area. The positive direction assumes no over-riding toxic effect of high pH in the porewater. Although diatom species will be preferentially promoted over other species, the current population is dominated by diatoms and will not change appreciably.

Zooplankton

Zooplankton species composition and biomass may change depending upon the interactions between predation and food availability, but impact magnitude is expected to be low

The response of the zooplankton community to increased phytoplankton biomass is difficult to predict. According to McQueen *et al*. (1986), the biomass of each trophic level is controlled from below by nutrient availability and bottom-up effects attenuate quickly along food chains. McQueen *et al*. (1986) suggest that the importance of top-down effects of predation increases in oligotrophic systems. However, other studies suggest that top-down effects of predation are weak in oligotrophic lakes (Johnstone *et al.* 1999). According to Johnston *et al*. (1999), the expected response in the north lake zooplankton would be a proportionate increase in biomass according to the increase in phytoplankton, with a very weak influence of predation by larval fish. A major shift in zooplankton community structure is not expected in the north lake because phytoplankton community structure is not expected to change from the current diatom-dominated community. Since the prey species assemblage is not expected to change, the grazer species assemblage is also not expected to change. The direct impact of an increase in phytoplankton on zooplankton biomass is positive in direction. An increase in phytoplankton (and productivity) will increase secondary productivity (zooplankton). Impact magnitude is expected to be low based upon a low increase in primary productivity in less than 10% of the lake area. The positive direction of impact assumes no over-riding toxic effect of high pH in the porewater. It also assumes no toxicity from the conversion of nitrate to un-ionized ammonia in the sediments. These assumptions may not hold true; however, overall effects of the combined chemical stressors plus pH is dealt with in Section 9.5.2.5 (Question F-4).

Benthos

The benthic invertebrate community is expected to be relatively insensitive to limited nutrient loading

A major change in benthic organism community structure is normally associated with gross organic pollution and/or sedimentation (Ward 1992). Nitrogen nutrient loading to the north lake (*i.e.*, nitrate sediment concentrations of 42.7 mg/L) is not sufficient to cause a substantial change in community structure because the baseline benthic community structure consists of taxa with a very broad tolerance to a range of nutrient conditions.

The biomass of the benthic invertebrate community may increase

The expected increases in nitrate concentrations and primary production in the north lake may lead to an increase in benthic invertebrate biomass, as observed by Johnston *et al*. (1999) during an experimental fertilization of an Arctic oligotrophic lake. The magnitude of the increase in biomass will depend upon the interaction between nutrient enrichment and predation.

The impact magnitude of an increase in primary production on benthos is positive and of low magnitude

Chironomidae dominate the regional lakes and are extremely adaptable; contributing a progressively greater proportion of the diversity and abundance of lake benthos from the sub-arctic to the high Arctic (Ward 1992). Chironomid tribes *Chironomini* and *Tanytarsini* compose the majority of chironomids in Snap Lake. *Tanytarsus* exhibit optimum growth in oligo-mesotrophic nutrient regimes and will tolerate meso-eutrophic nutrient levels to a lesser extent (Ward 1992). Chironomini respond similarly to the predicted nutrient regime. The primary limiting parameter of nematodes is oxygen availability (Wetzel 2001). Anoxic conditions directly limit nematode community composition and abundance. Dissolved oxygen concentrations are not likely to plummet in the winter due to decreased metabolic activity and may remain sufficient in the summer due to wind mixing and a shallow bathymetry. Nematode populations will not be substantially affected by low-level nutrient enrichment of the north lake sediments. The overall impact of an increase in nitrate is expected to be positive in direction (because of an increase in primary productivity). The impact magnitude is expected to be low, based upon low increase in primary productivity in less than 10% of lake area. The positive direction of impact assumes no over-riding toxic effect of high pH in the sediment. It also assumes no toxicity from the conversion of nitrate to un-ionized ammonia. These assumptions may hold true; however, the overall effects from the combined chemical stressors plus pH are dealt with in Section 9.5.2.5 (Question F-4).

Aluminum

The general guideline for the protection of aquatic life is 100 μg/L

Baseline aluminum in the north and northeast lakes averages <30 µg/L (detection limit). The proposed Snap Lake Diamond Project will release aluminum to the sediments of the north and northeast lakes. The maximum porewater concentration for aluminum will be 468 µg/L. The general guideline for the protection of aquatic life is 100 µg/L.

Aluminum speciation and toxicity varies with pH

In contrast to the amount of information concerning aluminum toxicity at low to mid pH ranges, the data concerning high pH aluminum toxicity is limited and qualitative (Morel and Hering 1993). Aluminum ion, Al^{3+}, is toxic and is normally bound as various $Al(OH)_n$ compounds at pHs between 6 and 9. Aluminum toxicity is normally associated with low pH environments, where more elemental ion (Al^{3+}) is available. The predicted sediment pH in the north and northeast lake sediments is 11.8. At this pH, essentially all aluminum is present as $Al(OH)_4^-$ and very limited $Al(OH)_3$ (Cooke *et al*. 1993, Morel and Hering 1993).

The aluminate ion is present and toxic at high pH

Aluminum hydroxide ($Al(OH)_{3-}$) has low or zero toxicity to aquatic biota (Cooke *et al*. 1993). However, the aluminate ion ($Al(OH)_4^-$) associated with higher pH (pH> 9 to 10) is toxic (Morel and Hering 1993). George *et al.* (1991) found the aluminate ion to be directly toxic to the algal species *Selenastrum capricornutum*. At a pH of 8 to 9, aluminum is toxic to *Daphnia* at concentrations of 100 µg/L to 300 µg/L (Mance 1990). Microcosm results indicate alum sludge deposits on sediment may detrimentally affect benthic macroinvertebrate populations by limiting their access to carbon or food (George *et al*. 1991). BC Environment (1988) determined a safe concentration of suspended aluminum in alkaline waters to be less than 0.5 mg/L total aluminum. They determined that a concentration of 0.05 mg/L total aluminum at pH 8 did not have toxic affects.

Phytoplankton, Zooplankton, and Benthos

The assessment of impact magnitude from aluminum is based upon very limited data

There are very few data effects of aluminum at very high pH. What little information is available does not provide quantitative toxicity thresholds; rather, qualitative descriptions such as "toxic" are applied to the aluminate ion (the prevalent form at high pH). Therefore, prediction of the magnitude of impact from aluminum on all three major groups on non-fish aquatic organisms is difficult.

The impact magnitude was estimated as low for phytoplankton, zooplankton, and benthos

The predicted porewater aluminum concentration of 468 g/L is well above the CCME guideline of 100 g/L; however, the CCME guideline is not based upon data for high pH ranges. Since the aluminate ion is referred to as "toxic" by Morel (1993), it is assumed that the predicted porewater concentration will exceed toxicity thresholds for all three major groups of non-fish aquatic organisms. These exceedances will be limited to less than 10% of the lake area, and will occur year-round. The overall magnitude of impact has been rated as low; however, this estimate has a high level of uncertainty.

9.5.2.2.4 Residual Impact Classification

Snap Lake: Construction and Operations

Negligible environmental consequences to non-fish aquatic organisms in Snap Lake are predicted

The residual impacts of the Snap Lake Diamond Project on the quantity and quality of non-fish aquatic organisms in Snap Lake are summarized in Table 9.5-21. Although any potential impacts to non-fish aquatic organisms would be long-term, they would be of negligible magnitude and reversible. Therefore, the environmental consequence for all potential impacts has been rated as low for Snap Lake.

Environmental consequences to the zooplankton community may be low because of effects of elevated calcium concentrations on sensitive cladoceran species

Environmental consequences to the zooplankton community are somewhat higher (though still in the low category) because of potential effects of elevated calcium concentrations on sensitive cladoceran species. Maximum predicted calcium concentrations may approach or slightly exceed chronic effect levels for cladocerans. These effects would be limited to less than 10% of the lake and would be seasonal (winter only).

Table 9.5-21 Residual Impact Classification for Changes to Non-Fish Aquatic Organisms in Snap Lake

Chemical of Concern	Trophic Level	Direction	Magnitude	Geographic Extent	Duration	Reversibility	Frequency	Environmental Consequences
Hexavalent chromium	phytoplankton	negative	negligible	local	long term	reversible (long-term)	high	low
Hexavalent chromium	zooplankton	negative	negligible	local	long term	reversible (long-term)	high	low
Hexavalent chromium	benthos	negative	negligible	local	long term	reversible (long-term)	high	low
Total dissolved solids	phytoplankton	positive	negligible	local	long term	reversible (long-term)	high	n/a
Total dissolved solids	zooplankton	negative	low[1]	local	long term	reversible (long-term)	high	low
Total dissolved solids	benthos	negative	negligible	local	long term	reversible (long-term)	high	Low

Note: Method used to determine environmental consequence is explained in Section 9.1.
[1.] Due to calcium ion
n/a = not applicable. The environmental consequence is not calculated for positive impacts.

Snap Lake: Post Closure

No residual impacts are predicted for the post-closure period for Snap Lake

All discharges to Snap Lake will cease in the post-closure period and the water quality will gradually return to baseline conditions (see Section 9.4). Negligible impacts were predicted from elevated chromium for the construction and operations phase; therefore, impacts in post-closure would also be negligible before completely disappearing once chromium concentrations return to baseline. Elevated calcium concentrations would also return to baseline levels during post closure, returning impacts on sensitive cladoceran species to negligible.

North Lake, Northeast Lake, NL5 and NL6: Post-Closure

Residual impacts during post closure range from negligible to moderate depending upon the chemical of concern

The residual impacts of the Snap Lake Diamond Project on the quantity and quality of non-fish aquatic organisms in NL5 and NL6, the north lake, and the northeast lake are summarized in Tables 9.5-22, 9.5-23, and 9.5-24 respectively. Consequences in NL5 and NL6 are from increases in water column concentrations of trivalent chromium only. Consequences in the north and northeast lakes are from increases in trivalent chromium in the water column of the north lake, and increases in porewater concentrations of trivalent and hexavalent chromium, aluminum, copper, molybdenum, pH and nitrates.

All water column post closure environmental consequences are low

The environmental consequences of an increase in trivalent chromium concentrations in the water column of the north lake, NL5 and NL6 are low. This is because of the negligible magnitude of impact on phytoplankton, zooplankton and benthos.

Most sediment porewater post closure environmental consequences are low

The environmental consequences of increases in trivalent chromium, copper and aluminum in the sediment porewater of the north and northeast lakes are uniformly low. This is true across all three major groups of non-fish aquatic organisms. The low consequences are driven by the low-to-negligible magnitude of impact for the three metals.

Moderate environmental consequences are predicted for hexavalent chromium and pH in sediment porewater

Hexavalent chromium and pH are predicted to cause moderate environmental consequences in the north and northeast lakes. This is true for phytoplankton, zooplankton, and benthos. The moderate impacts from chromium are predicted based upon exceedance of chronic toxicity thresholds. The moderate impacts from pH are predicted because of the potential for acutely lethal effects in up to 10% of the north and northeast lake sediments.

Nitrate impacts are expected to be positive in direction; therefore, environmental consequences are not rated

Increased nitrate concentrations in the sediment porewater of the north and northeast lakes are expected to cause low magnitude increases in primary and secondary productivity. Because of the positive direction of impacts, environmental consequences are not rated. The positive direction of impacts assumes that there will be no over-riding toxic effects from the high pH in the sediments. This is an over-simplification; however, the overall effects of the combination of chemical stressors and pH are evaluated in Section 9.5.2.5 (Key Question F-4).

Table 9.5-22 Residual Impact Classification for Changes to Non-Fish Aquatic Organisms in NL5 and NL6[1], Water Column Only

Chemical of Concern	Physical Level	Trophic Level	Dir.	Magnitude	Geographic Extent	Duration	Reversibility	Freq.	Environmental Consequences
Trivalent chromium	water	phytoplankton	neg.	negligible	local	long-term	reversible (long-term)	high	low
Trivalent chromium	water	zooplankton	neg.	negligible	local	long-term	reversible (long-term)	high	low
Trivalent chromium	water	benthos	neg.	negligible	local	long-term	reversible (long-term)	high	Low

Note: Numerical score for ranking of environmental consequence is explained in Section 9.1.

[1] NL5 and NL6: Trivalent chromium in water column only.

Table 9.5-23 Residual Impact Classification for Changes to Non-fish Aquatic Organisms in the North Lake

Chemical of Concern	Physical Level	Trophic Level	Dir.	Magnitude	Geographic Extent	Duration	Reversibility	Freq.	Environmental Consequences
Trivalent chromium	water	phytoplankton	neg.	negligible	local	long-term	reversible (long-term)	high	low
Trivalent chromium	water	zooplankton	neg.	negligible	local	long-term	reversible (long-term)	high	low
Trivalent chromium	water	benthos	neg.	negligible	local	long-term	reversible (long-term)	high	low
Trivalent chromium	sediment	phytoplankton	neg.	negligible	local	long-term	reversible (long-term)	high	low
Trivalent chromium	sediment	zooplankton	neg.	negligible	local	long-term	reversible (long-term)	high	low
Trivalent chromium	sediment	benthos	neg.	negligible	local	long-term	reversible (long-term)	high	low
Hexavalent chromium	sediment	phytoplankton	neg.	moderate	local	long-term	reversible (long-term)	high	moderate
Hexavalent chromium	sediment	zooplankton	neg.	moderate	local	long-term	reversible (long-term)	high	moderate
Hexavalent chromium	sediment	benthos	neg.	moderate	local	long-term	reversible (long-term)	high	moderate
Copper	sediment	phytoplankton	neg.	low	local	long-term	reversible (long-term)	high	low
Copper	sediment	zooplankton	neg.	low	local	long-term	reversible (long-term)	high	low
Copper	sediment	benthos	neg.	low	local	long-term	reversible (long-term)	high	low
Molybdenum	sediment	phytoplankton	neg.	low	local	long-term	reversible (long-term)	high	low
Molybdenum	sediment	zooplankton	neg.	negligible	local	long-term	reversible (long-term)	high	low
Molybdenum	sediment	benthos	neg.	negligible	local	long-term	reversible (long-term)	high	low
pH	sediment	phytoplankton	neg.	moderate	local	long-term	reversible (long-term)	high	moderate
pH	sediment	zooplankton	neg.	moderate	local	long-term	reversible (long-term)	high	moderate
pH	sediment	benthos	neg.	moderate	local	long-term	reversible (long-term)	high	moderate
Nitrate	sediment	phytoplankton	pos.	moderate	local	long-term	reversible (long-term)	high	n/a
Nitrate	sediment	zooplankton	pos.	low	local	long-term	reversible (long-term)	high	n/a
Nitrate	sediment	benthos	pos.	low	local	long-term	reversible (long-term)	high	n/a
Aluminum	sediment	phytoplankton	neg.	low	local	long-term	reversible (long-term)	high	low
Aluminum	sediment	zooplankton	neg.	low	local	long-term	reversible (long-term)	high	low
Aluminum	sediment	benthos	neg.	low	local	long-term	reversible (long-term)	high	low

Note: Method used to determine environmental consequence is explained in Section 9.1.
a = not applicable. The environmental consequence is not calculated for positive impacts.

Table 9.5-24 Residual Impact Classification for Changes to Non-Fish Aquatic Organisms in the North Lake

Chemical of Concern	Physical Level	Trophic Level	Dir.	Magnitude	Geographic Extent	Duration	Reversibility	Freq.	Environmental Consequences
Trivalent Chromium	sediment	phytoplankton	neg.	negligible	local	long-term	reversible (long-term)	high	low
Trivalent Chromium	sediment	zooplankton	neg.	negligible	local	long-term	reversible (long-term)	high	low
Trivalent Chromium	sediment	benthos	neg.	negligible	local	long-term	reversible (long-term)	high	low
Hexavalent chromium	sediment	phytoplankton	neg.	moderate	local	long-term	reversible (long-term)	high	moderate
Hexavalent Chromium	sediment	zooplankton	neg.	moderate	local	long-term	reversible (long-term)	high	moderate
Hexavalent Chromium	sediment	benthos	neg.	moderate	local	long-term	reversible (long-term)	high	moderate
Copper	sediment	phytoplankton	neg.	low	local	long-term	reversible (long-term)	high	low
Copper	sediment	zooplankton	neg.	low	local	long-term	reversible (long-term)	high	low
Copper	sediment	benthos	neg.	low	local	long-term	reversible (long-term)	high	low
Molybdenum	sediment	phytoplankton	neg.	negligible	local	long-term	reversible (long-term)	high	low
Molybdenum	sediment	zooplankton	neg.	negligible	local	long-term	reversible (long-term)	high	low
Molybdenum	sediment	benthos	neg.	negligible	local	long-term	reversible (long-term)	high	low
pH	sediment	phytoplankton	neg.	moderate	local	long-term	reversible (long-term)	high	moderate
pH	sediment	zooplankton	neg.	moderate	local	long-term	reversible (long-term)	high	moderate
pH	sediment	benthos	neg.	moderate	local	long-term	reversible (long-term)	high	moderate
Nitrate	sediment	phytoplankton	pos.	moderate	local	long-term	reversible (long-term)	high	n/a
Nitrate	sediment	zooplankton	pos.	low	local	long-term	reversible (long-term)	high	n/a
Nitrate	sediment	benthos	pos.	low	local	long-term	reversible (long-term)	high	n/a
Aluminum	sediment	phytoplankton	neg.	low	local	long-term	reversible (long-term)	high	low
Aluminum	sediment	zooplankton	neg.	low	local	long-term	reversible (long-term)	high	low
Aluminum	sediment	benthos	neg.	low	local	long-term	reversible (long-term)	high	low

Note: Method used to determine environmental consequence is explained in Section 9.1.
n/a = not applicable. The environmental consequence is not calculated for positive impacts.

9.5.2.2.5 Level of Certainty in the Impact Assessment

There is a high level of certainty in the residual impact predictions for Snap Lake during construction, operations, and post closure

Residual impact predictions for Snap Lake are based upon conservative, but well-calibrated water quality predictions and knowledge of baseline conditions. The conservative nature of the water quality predictions means that the risk of under-estimating the environmental consequences in Snap Lake is low. Furthermore, the knowledge of baseline conditions increases the certainty that the interpretation of predicted water quality effects on the aquatic biota was relevant and defensible.

There is a low level of certainty in the residual impact predictions for the north, northeast, NL5, and NL6 lakes during post closure; however, conservative assumptions are part of the assessment

Residual impact predictions for the post closure case conditions in the north, northeast, NL5 and NL6 lakes are uncertain. There are several sources of uncertainty. Water quality predictions are based upon the groundwater modelling. The groundwater model is uncertain because of the difficulty in predicting geochemical conditions, the lack of calibration data, and the uncertainty in the identification and characterization of groundwater flow paths. The water quality model is uncertain because of the lack of baseline data for use in calibration; however, this uncertainty is dealt with by using conservative assumptions of both groundwater flow volumes and concentrations of chemicals of concern. Water quality in NL5 and NL6 was not modelled. The effects assessment is uncertain because of the lack of baseline data on the aquatic communities in the four lakes. Again, to deal with the lack of baseline data, conservative assumptions regarding the types of organisms expected to be present and the potential for exposure were factored into the assessment of impact magnitudes.

9.5.2.3 Key Question F-2: What Impacts Will the Snap Lake Diamond Project Have on Fish Habitat?

9.5.2.3.1 Linkage Analysis

Seven linkages were analyzed

The following potential linkages between the Snap Lake Diamond Project and fish habitat were analyzed:

- linkage between blasting activity and changes to fish habitat;
- linkage between construction or removal of in-stream structures and changes to fish habitat;
- linkage between the construction of stream crossings and changes to fish habitat;
- linkage between development of seepage and runoff collection ponds and changes to fish habitat;
- linkage between dust deposition and changes to fish habitat;

- linkage between changes in hydrology and changes to fish habitat; and,
- linkage between changes to non-fish aquatic organisms and changes to fish habitat.

Blasting Activity

The pathway between blasting activity and fish habitat change is invalid

Blasting will be used as part of the underground mining process as well as during quarrying activities on the northwest peninsula. The potential impacts from the use of explosives in or near fish habitat include physical or chemical alteration of the habitat (Wright and Hopky 1998). Physical changes would be the result of sediment release that may cover spawning areas or eliminate benthic invertebrate food sources. Chemical changes may result from the release of explosive by-products such as ammonia (Wright and Hopky 1998). Blasting in relation to the proposed Snap Lake Diamond Project will not occur within any surface waterbodies. Blasting associated with mining activity will be conducted underground and mine water, with the associated sediment and chemical by-products, will be pumped to the water treatment plant. The treatment and release of this water is evaluated and discussed in Section 9.4 of this report. Quarrying activities will occur in designated areas of the northeast peninsula (see Section 3.7.3 and Appendix III.5). These areas are within the runoff water containment system surrounding the mine footprint so that water carrying sediment or chemical by-products will be diverted to seepage and runoff collection ponds and not released directly to Snap Lake. The treatment and release of this water is also evaluated and discussed in the Water Quality section of this report (Section 9.4). Based on this information, the pathway from mine blasting activity directly to fish habitat is not a valid pathway.

Construction/Removal of Instream Structures

Water intake and outlet structures are proposed for construction in Snap Lake

The project description (Section 3) identifies two structures to be constructed within Snap Lake. These are the water intake structure and waste water outlet structure. The water intake structure is proposed for construction within Snap Lake, on the northeast peninsula (see Section 3.6 and Figure 3.1-4). This structure will cover approximately 42 m^2 (0.0042 ha) of lake bottom and affect 787 m^2 of deepwater habitat.

The water outlet structure includes a rock filled embankment, a pipeline, and a diffuser

The water outlet structure includes three main components. The construction of a rock filled embankment is proposed at the shoreline to ensure the pipeline is protected from ice, wind, and wave action. There will also be an insulated pipeline extending approximately 125 m out from shore to a depth of 12 m. At the end of the pipeline, there will be 70 m long diffuser structure with seven evenly spaced outlet ports. This structure will affect approximately 760 m^2 (0.0760 ha) of lake bottom and alter 0.0098 m^2 of deep water habitat (see Section 3.6 and Figure 3.1-4).

The linkage between instream structures and a change in fish habitat is a valid pathway

Based on the construction of two separate structures within Snap Lake, and the associated changes to lake bottom at these two locations, the pathway from construction and/or removal of instream (*i.e.*, within a waterbody) structures to a change in fish habitat is a valid pathway.

Stream Crossings

Changes to fish habitat as a result of stream crossings was considered

The potential for stream alteration and sediment deposition was considered with respect to development of roads and other infrastructure for the proposed project in the Snap Lake drainage basin.

Specifically, effects on fish habitat may result from:

- direct disturbance, alteration, or loss of productive habitats at the watercourse crossing site; and,
- increased deposition of fine sediments downstream of the crossing site resulting from instream construction activities or runoff from newly excavated banks or approach slopes.

The linkage between stream crossings and a change in fish habitat is not a valid pathway

One watercourse crossing was identified within the project footprint. Stream S31 (Figure 9.5-3) has been crossed by the construction of the airstrip. This crossing was previously permitted as part of the Class A Water License and, based on the Terms of Reference, will therefore not be evaluated further. No other stream crossings occur within the project footprint, therefore the linkage between stream crossings and a change to fish habitat is not a valid pathway.

Development of Seepage and Runoff Collection Ponds

The linkage between seepage and runoff collection ponds and changes to fish habitat in the inland lakes invalid

Three inland lakes, IL6, IL7, and IL9 (Figure 9.5-3) found within the proposed footprint of the north pile have been identified as suitable for development as seepage and runoff collection ponds. The use of a lake as a sedimentation pond would lead to several physical and chemical habitat changes including water level fluctuation, deposition of sediment on rocky substrates, increases in TSS, and alteration of parameters such as TDS. None of these inland lakes were found to provide fish habitat and stream channels do not connect them with any other waterbodies found to provide fish habitat (Section 9.5.1.5). Based on this, the linkage between the development of seepage and runoff collection ponds and changes to fish habitat in the inland lakes is not a valid pathway.

There is no linkage between seepage and runoff collection ponds and fish habitat in Snap Lake

In addition, water from the seepage and runoff collection ponds will flow within the mine water containment system during construction and operations and therefore will not be released directly to Snap Lake. As a result there is also no linkage between development of seepage and runoff collection ponds and effects to fish habitat in Snap Lake.

Dust Deposition

The linkage between increased dust deposition and fish habitat is a valid

There are a number of activities associated with mine construction and operation that could contribute to elevated dust emissions from the Snap Lake Diamond Project. These in turn could affect fish habitat. These activities include power generation, underground mining, waste incineration, dust from road use and particles transported by wind action off the north pile and the quarry. Dust emissions or total suspended particulates (TSP) could potentially result in accumulations of sediment on lake bottoms altering fish habitat, thereby affecting near-shore spawning, rearing, foraging or refuge areas. Thresholds have been identified that indicated that an accumulation greater than 1 mm is sufficient to cause decreased fish egg survival (Fudge and Bodaly 1984). Based on this, the linkage between increased dust deposition and changes in fish habitat is valid.

Change to Hydrology

The linkage between lake levels and fish habitat in Snap Lake is invalid

Water withdrawals and water releases to Snap Lake during construction and operations have the potential to contribute to a change in lake water levels. A change in water levels could cause the loss of near-shore spawning, rearing, foraging, or refuge areas for a number of fish species. A change in water levels also has the potential to alter water depths of other areas of fish habitat and alter the quality of these areas for some fish. However, the potential change to lake levels in Snap Lake was evaluated in Hydrology (Section 9.3.2). Since the change in lake levels in Snap Lake were assessed as having a negligible environmental consequence, the linkage between changes to lake levels and the potential for changes to fish habitat in Snap Lake is invalid.

The linkage between lake levels and fish habitat in other local study area lakes is invalid

This linkage between lake levels and fish habitat was also evaluated for the north and northeast lakes (located north and north east of Snap Lake), and NL5 and NL6 (Figure 9.5-1). Lake levels in these lakes are predicted to be negligibly lower, and have a negligible environmental consequence, as a result of reduced groundwater flow (Section 9.3.2). Therefore, this linkage is invalid.

The linkage between water levels and fish habitat in the inland lakes is invalid

Water level changes have the potential to affect fish habitat in any inland lakes providing fish habitat. Water level changes may be due to the development of sedimentation ponds, changes in sub-basin water flow, and direct water removal from these lakes. Lake levels could remain altered if sub-basin re-contouring post-closure does not reflect baseline conditions. Lakes with the possibility of these types of impacts occurring include IL6, IL7, and IL9. These lakes were found to not provide fish habitat and they were also not connected with any other waterbodies found to provide fish habitat (Section 9.5.1.5). Thus, the linkage between changes to water levels and fish habitat in the inland lakes is not a valid pathway.

Changes to fish habitat, as a result of increased annual flow, in outlet streams from Snap Lake is invalid

Lake level changes may also affect outlet streams. Increased outflow from Snap Lake was evaluated as having a negligible environmental consequence (Section 9.3.2) as the change was seasonal, well within natural variability, and would not result in a change to channel morphology. Based on this, the linkage between a change in fish habitat in the outlet streams from Snap Lake, and a change in hydrology, is not valid.

Changes to fish habitat, as a result of reduced annual flow, in the outlet stream is a valid linkage for the north lake

Lower lake levels are predicted to affect the volume of flow out of the north lake and the northeast lake (Section 9.3.2). These changes in flow have the potential to disrupt fish habitat in the outlet streams from these lakes. The reduction in flow was evaluated as low for the outlet from the north lake and negligible for the outlet from the northeast lake (Section 9.3.2). Based on this information, the linkage between changes to annual flow and the potential for changes to fish habitat in the outlet stream is a valid linkage for the north lake but not for the northeast lake.

Potential effects to fish habitat through alteration of runoff in sub-basins S and Q is a valid linkage

Hydrologic alteration of sub-basin watersheds, as a result of mine construction and operation, may affect delivery of water to stream channels. Fish habitat in these streams may be affected through changes to the volume of flows, persistence or duration of flows, and timing of flows. Sub-basins overlapping the mine footprint include: A, C, D, E, F, G, H, I, J, K, L, M, N, O, P, Q, R, and S. Of these basins, S and Q were identified as having stream channels that provide fish habitat (Section 9.5.1.5). As a result, potential effects to fish habitat through alteration of runoff is a valid linkage.

The linkage between sediment yield and changes to fish habitat is valid for Snap Lake and streams within-sub-basins S and Q

Alteration of the hydrological regime in the sub-basins, due to watershed disturbance during construction and operation of the Snap Lake Diamond Project, could have an affect on the sediment yield delivered to water bodies associated with the project. As a result, there is the potential for an affect on fish habitat. Also the development of the WMP could result in increased sediments entering the surface water environment of the northwest peninsula. Increased sediment loads entering Snap Lake could alter near-shore spawning, rearing, foraging, or refuge areas for a number of fish species. Similarly, changes to fish habitat are possible in the two streams associated with sub-basins S and Q if sediment loading were to increase. As discussed above, IL6, IL7, and IL9, found within the project footprint, were not found to provide fish habitat and they are also not connected with any other waterbodies found to provide fish habitat. Based on this information the linkage between sediment yield and changes to fish habitat is valid for Snap Lake and streams within sub-basins S and Q.

Change to Non-fish Aquatic Organisms

The linkage between non-fish aquatic organisms and fish habitat is valid for the north lakes and valid for calcium only in Snap Lake

Indirect effects on fish habitat may occur because of changes in the abundance or quality of non-fish aquatic organisms that are consumed by fish. This is based on the definition of fish habitat in the *Fisheries Act*, which includes food and all other resources required by fish of all life stages. The residual impact classification for Key Question F1 (Section 9.5.2.2) indicated a negligible impact magnitude and environmental consequence in relation to changes to non-fish aquatic organisms in Snap Lake with the exception of calcium. An environmental consequence of low was determined for Snap Lake zooplankton in relation to the potential for chronic effects to some species. Several low and moderate ratings for environmental consequence are predicted for the north lake, NL5, NL6, and the north east lake. Based on this, the linkage between a change in non-fish aquatic organisms and an effect on fish habitat is valid for these LSA lakes.

9.5.2.3.2 Mitigation

Four linkages were valid

Based on the analysis of the potential linkage between project activities and fish habitat changes, four pathways were identified as valid. These were construction and removal of instream structures, dust deposition, changes to hydrology, and changes to non-fish aquatic organisms.

Construction and Removal of Instream Structures

The water intake was designed to use the rock fill structure as a screen and avoid effects to fish

The water intake was designed to avoid the need for fish screens by building a rock filled embankment constructed out from the shore. The embankment will be constructed around vertical filtration wells fitted with vertical turbine pumps to supply water on demand. As indicated in the project description, the water intake will not incorporate any screens, as the rock filled embankment will perform the screening function. This will also minimize maintenance requirements. In addition, the intake structure has been designed to meet the DFO guidelines for intake velocities (DFO 1995).

Sediment release will be controlled during construction

Construction of both the water intake and outlet structures will be conducted to minimize the release of sediment to Snap Lake. The fill material will be non-PAG, 600 mm minus washed rock. Silt curtains or other sediment control technology will be implemented as needed to meet any applicable regulations. Erosion from any overland flow will also be controlled adjacent to the intake construction area.

Rock size on the outer surface will be sized to provide fish habitat

Sizing of rock on the outer surface of both structures will be configured to both protect them from physical damage and erosion forces, as well as provide a measure of fish habitat. Crushed, clean, non-PAG rock sized from 10 to 25 cm will be used.

Following the construction of these structures, very little additional habitat alteration will occur

Following construction, the intake and outlet structures will be in operation for the life of the mine. At closure the intake structure will be partially removed to allow for the reclamation of the pumping apparatus, but much of the rock fill will be left in place (increasing available habitat) and no further alteration will occur. At the outlet, the diffuser structure and pipeline will be removed but the rock embankment at the base of the pipeline will remain. Habitat alteration at this stage will be minimized by following best practices for instream (in-water) activity.

Dust Deposition

Wet processes and contained activities are key mitigation approaches

Mitigation procedures for TSP include the following:

- the dust suppression program for roads and the airstrip;
- the primary crushing conducted as an underground operation as well as any other crushing being conducted under wet processing conditions;
- the high moisture content of materials placed in the north pile;
- the addition of water to the surface of the north pile for dust suppression; and,
- the capping of the north pile at closure.

In addition, any of the dry operations, such as the addition of cement powder to the spent kimberlite, have effective dust control systems in place.

Change to Hydrology

The installation of culverts for cross drains, re-contouring following closure, development of the seepage runoff and collection ponds, and an appropriate water balance will all mitigate changes to hydrology

Mitigation for changes to hydrology include several project design considerations. Culverts will be installed to provide cross drainage along roadways and the airstrip where necessary. This will provide for continued flow following the current drainage pattern. Following mine closure, the natural drainage areas and flow directions will be reestablished to baseline conditions in the mine footprint area. This will mitigate changes to stream flows in the LSA. In addition, the development of the seepage and runoff collection ponds will mitigate potential increases in sediment yield to Snap Lake. Changes to lake levels in Snap Lake will be mitigated by maintaining an appropriate water balance between water withdrawal and water release.

Change to Non-fish Aquatic Organisms

Mitigation for potential impacts to non-fish aquatic organisms are discussed in Water Quality (Section 9.4)

Potential impacts of the Snap Lake Diamond Project on non-fish aquatic organisms are directly related to the collection, treatment, and release of contaminated water on site. Mitigation plans have been discussed in the water quality assessment (Section 9.4) and the WMP (Appendix III.4). The measures discussed in those sections will reduce potential impacts to non-fish aquatic organisms. No specific mitigation measures are discussed in this section.

9.5.2.3.3 Impact Analysis

Construction and Removal of Instream Structures

A modified Habitat Evaluation Procedure was used to calculate the quantity and quality of fish habitats being altered, lost or created

A modified HEP was used to calculate the quantity and quality of fish habitats being altered, lost, or created during all phases of the proposed project (*i.e.*, baseline, construction, operations, and post-closure) (USFWS 1981). HEP analysis combines the habitat quality, defined as HSI, with habitat quantity to calculate HUs. HUs provide a measure that accounts for both the quantity and quality of habitat available for certain species. Multiplying the suitability rating (HSI value) by the area of habitat affected provides the number of HUs available.

Comparing the number of habitat units available under baseline conditions to those available during the various project phases allows for the quantification of project effects

Comparing the number of HUs available under baseline conditions to those available during construction, operations, and post-closure allows for the quantification of the overall number of HUs altered, lost, and created by the proposed project. Thus, the difference in habitat quantity, weighted by habitat quality, between existing and future conditions enables an evaluation of the net effect of the proposed project on fish habitat. The HEP approach has been used to assess habitat requirements of fish species associated with other diamond projects in the NWT and is further described in Diavik (1998a and 1998b).

The areas of impact to aquatic habitat were evaluated for nearshore (<4 m), deep water (>4 m), and shoal environments combined to represent 10 habitat types

The areas of impact to aquatic habitat associated with the development and operation of the Snap Lake Diamond Project were evaluated for three main environments found in Snap Lake. These included, near shore (<4 m), deep water (>4 m), and shoal habitat (0 – 6 m). Within these environments, habitat was subdivided into distinct types (Table 9.5-9) and categorized in relation to its suitability to fish in various life stages (*i.e.*, spawning, rearing, foraging, and nursery). Habitat types were not exclusive to one species or life stage so that each individual physical environment and habitat type could be used for more than one habitat requirement. For example, nearshore boulder and cobble habitat was assessed as providing both spawning and rearing habitat of varying values for particular species.

The area of nearshore and deep water habitat affected by the water intake and outlet structures was calculated

Both the water intake and mine water outlet structure footprints were considered in the calculation of habitat lost or altered. The water intake structure will consist of a rock filled embankment constructed out from shore to access water at a depth of 7 m. The pad will cover 0.0042 ha of lake bottom on the north side of the northeast peninsula affecting boulder cobble and 0.0787 ha of deep water habitat. The mine water outlet structure will consist of an embankment structure covering 0.076 ha of lake bottom (boulder/cobble habitat), and an insulated pipe running out from the south side of the northeast peninsula to a diffuser structure on the lake bottom. The pipeline and diffuser will alter 0.0160 ha of deep water habitat.

Appendix IX.12 provides a full explanation and summary of calculations used to determine habitat units in Snap Lake

Appendix IX.12 provides a full explanation and summary of calculations used to determine HUs in Snap Lake. The appendix lists the HSI used, as well as the weightings applied following the defensible methods approach (Minns 1995). Details of the mitigation plans, structure designs, and structure removal plans will be provided in a Fish Habitat Compensation Plan to be submitted to DFO.

Table 9.5-25 summarizes the habitat lost to the instream structures

Table 9.5-25 summarizes the habitat lost to the water intake and the mine water diffuser structures and compares the areas to the available habitat in all of Snap Lake and the northwest peninsula. These areas were used in the calculations of HUs for the impact analysis.

The habitat types created by mitigation at the water intake and mine water outlet were also calculated

The areas of various habitat types created by the construction of the water intake and mine water outlet structures were calculated based on the size and materials used in the construction. The outer surface of the water intake structure will be constructed using rock that will simulate boulder/cobble habitat and will result in 0.0392 ha of new habitat to the 4 m contour. The water intake structure will also extend below 4 m and create 0.0705 ha of new boulder/cobble habitat from 4 to 7 m contours. This deeper boulder/cobble habitat was classified as a new habitat type. The pad for the water outlet structure will be constructed out of rock that will also simulate boulder/cobble habitat and result in the creation of 0.0471 ha of habitat. A total of 0.1568 ha of habitat will be created by mitigation.

Table 9.5-25 Habitat Areas Lost to the Water Intake and Mine Water Outlet Structures in Relation to Snap Lake and the Northeast Peninsula

Habitat Type [1]	Habitat #	Habitat Area (ha) Water Intake (% of available)	Habitat Area (ha) Treated Water Diffuser (% of available)	Total Habitat Lost (% of available)	Total Habitat Area (ha) Snap Lake	Habitat Area (ha) Northeast Peninsula (% of available)
Nearshore Habitat (waters edge to 4 m contour)						
(Bo/Co)	1	0.0042 (0.0014)	0.0760 (0.0262)	0.0802 (0.0276)	290.48	52.40 (18.03)
Deep Water (> 4 m in depth)						
Deep water	8	0.0787 (0.0285)	0.0160 (0.0058)[2]	0.0947 (0.0343)	276.36	n/a

1 Bd, Bd/Bo, Bo, IV/Bo, Bd/Co, IV, primary and secondary shoal habitat are not affected by the construction of the instream structures.

2 This deep water habitat reclaimed at closure with the removal of the diffuser and pipeline.

Note: ha = hectare; Bo = boulder; Co = cobble; Bd = bedrock; IV = inundated vegetation.

HUs are used to predict potential habitat losses and gains by construction, mitigation or reclamation activities

In order to calculate HUs each type of habitat was assigned a numerical ranking of suitability for each species based off the HSIs (Appendix IX.12). The area, in hectares, of each habitat type were multiplied by the appropriate HSI values to obtain HUs. Once the HUs were calculated, they were then used to predict potential habitat losses and gains for each species caused by the development of instream structures in support of the proposed project.

Species weightings were applied and weighted habitat units were calculated for baseline and post-closure conditions

Species weightings derived from the defensible methods approach to assessing habitat loss (Minns 1995) and utilized in the Diavik Diamond Project No Net Loss Plan (Diavik 1999) were applied to the Snap Lake Diamond Project (Table 9.5-26). With this approach, weighted habitat units were calculated for baseline, construction, operations, and post-closure conditions. By applying weightings the relative importance of fish species in Snap Lake can be considered. Exploitation and species abundance form the basis of these weightings. Details can be found in Appendix IX.12.

Table 9.5-26 Exploitation and Abundance Weightings for Fish Species in Snap Lake

Species	Exploitation Weighting	Abundance Weighting	Final Weighting
LKTR	0.33	0.31	0.32
ARGR	0.2	0.03	0.11
RNWH	0.13	0.12	0.12
LNSC	0.07	0.05	0.06
BURB	0.13	0.00	0.07
SLSC	0.07	0.00	0.04
LKCH	0.07	0.50	0.28

Note: LKTR = lake trout; ARGR = Arctic grayling; RNWH = round whitefish; LNSC = longnose sucker; BURB = burbot; SLSC = slimy sculpin; LKCH = lake chub.

The net change in habitat units was calculated for the construction, operations, and closure phases

The net change in HUs for all life stages and each fish species were determined using the weighted HUs. The results for the construction, operations, and post-closure phases are shown in Tables 9.5-27 and 9.5-28. The percentage change in habitat was calculated relative to the amount of similar habitat available under baseline conditions. The assessment of the effects during the construction and operations phases was based on the proposed maximum mine development. The residual effect of the closure footprint was based on the conservative estimate of no change in the habitat created with the water intake structure, no change to rock embankment at the base of the diffuser pipeline, but the removal of the mine water diffuser structure and pipeline.

An overall gain of 0.0015 habitat units, during construction and operations, and 0.0258 habitat units, at closure, is predicted

An overall gain of 0.0015 HUs is predicted during construction and operations. This is expected to increase to 0.0258 HUs post-closure (Tables 9.5-27 and 9.5-28). When all life stages for each species are combined small gains in HUs are predicted during both construction and operations and post-closure phases for lake trout and lake chub (Table 9.5-29). Small losses in HUs are predicted for all other species.

Section 7.3.4 for Air Quality there will be negligible amounts of dust deposited on fish habitat in Snap Lake

Based on the impact assessment in Section 7.3.4 for Air Quality there will be negligible amounts of dust or TSP deposition on fish habitat. This is based on a review of deposition rates from activities associated with the Snap Lake Diamond Project construction and operations. Mining has been identified as the largest potential source of TSP, but as the activity occurs underground and the conditions are wet, TSP emissions are mitigated. Surface equipment and activities such as combustion engines, dust suppression measures on roads and laydown areas, and capping of the north pile will contribute the bulk of the TSP associated with the project. TSP deposition is predicted to be above guidelines for the area immediately associated with the project footprint, but the affect decreases substantially as distance from the project increases (see Section 7.3.4). For example, the annual TSP deposition is predicted to be

101.8 μg/m^3 in the RSA. When the area of the active mine site is eliminated from the calculation, the TSP drops to 18.0 μg/m^3. The criteria for acceptable limits annually is 60 μg/m^3 (Table 7.3-32).

Table 9.5-27 Net Change in Habitat Units Between Baseline and Construction and Operations with Weightings Applied

Life Stage	Species	Habitat Units (HUs)			% Change for Snap Lake
		[1]Baseline (Pre 2003)	Construction/ Operations (2003-2028)	Net Change	
Spawning	LKTR	31.5939	31.6000	0.0061	0.0193
	ARGR	0.0000	0.0000	0.0000	0.0000
	RNWH	12.6974	12.6998	0.0024	0.0187
	LNSC	0.0000	0.0000	0.0000	0.0000
	BURB	17.1992	17.1990	-0.0002	-0.0012
	SLSC	22.3866	22.3834	-0.0032	-0.0144
	LKCH	96.7500	96.7617	0.0117	0.0121
Rearing	LKTR	128.6847	128.6823	-0.0024	-0.0019
	ARGR	53.9152	53.9117	-0.0035	-0.0065
	RNWH	47.4222	47.4128	-0.0094	-0.0198
	LNSC	27.4201	27.4167	-0.0034	-0.0122
	BURB	36.7118	36.7070	-0.0048	-0.0131
	SLSC	23.7326	23.7300	-0.0026	-0.0109
	LKCH	151.7580	151.7563	-0.0017	-0.0011
Foraging	LKTR	174.2151	174.2169	0.0018	0.0010
	ARGR	46.5227	46.5190	-0.0037	-0.0079
	RNWH	67.1512	67.1444	-0.0068	-0.0101
	LNSC	27.4201	27.4167	-0.0034	-0.0122
	BURB	40.9848	40.9813	-0.0035	-0.0086
	SLSC	14.9223	14.9211	-0.0012	-0.0080
	LKCH	151.7580	151.7662	0.0083	0.0055
Nursery	LKTR	28.9962	29.0023	0.0061	0.0210
	ARGR	0.0000	0.0000	0.0000	0.0000
	RNWH	12.6974	12.6998	0.0024	0.0187
	LNSC	0.0000	0.0000	0.0000	0.0000
	BURB	17.1992	17.2032	0.0040	0.0234
	SLSC	22.3866	22.3834	-0.0032	-0.0144
	LKCH	96.7500	96.7617	0.0117	0.0121
Total		**1351.2754**	**1351.2769**	**0.0015**	**-0.0104[2]**

1 HUs available in the pre-development phase represent the number of HUs present on shoals, shorelines, and in deep water areas of Snap Lake.

2 Sum of % change column.

Note: LKTR = lake trout; ARGR = Arctic grayling; RNWH = round whitefish; LNSC = longnose sucker; BURB = burbot; SLSC = slimy sculpin; LKCH = lake chub.

Table 9.5-28 Net Change in Habitat Units Between Baseline and Post Closure Periods With Weightings Applied

Life Stage	Species	Habitat Units (HUs)			% Change for Snap Lake
		[1]Baseline (Pre 2003)	[2] Closure (2028)	Net Change	
Spawning	LKTR	31.5939	31.6000	0.0061	0.0193
	ARGR	0.0000	0.0000	0.0000	0.0000
	RNWH	12.6974	12.6998	0.0024	0.0187
	LNSC	0.0000	0.0000	0.0000	0.0000
	BURB	17.1992	17.1993	0.0001	0.0004
	SLSC	22.3866	22.3839	-0.0026	-0.0118
	LKCH	96.7500	96.7617	0.0117	0.0121
Rearing	LKTR	128.6847	128.6848	0.0002	0.0001
	ARGR	53.9152	53.9130	-0.0022	-0.0040
	RNWH	47.4222	47.4148	-0.0074	-0.0156
	LNSC	27.4201	27.4177	-0.0024	-0.0088
	BURB	36.7118	36.7081	-0.0037	-0.0102
	SLSC	23.7326	23.7306	-0.0020	-0.0084
	LKCH	151.7580	151.7585	0.0006	0.0004
Foraging	LKTR	174.2151	174.2207	0.0056	0.0032
	ARGR	46.5227	46.5204	-0.0023	-0.0050
	RNWH	67.1512	67.1464	-0.0048	-0.0072
	LNSC	27.4201	27.4177	-0.0024	-0.0088
	BURB	40.9848	40.9824	-0.0025	-0.0060
	SLSC	14.9223	14.9215	-0.0008	-0.0050
	LKCH	151.7580	151.7685	0.0105	0.0069
Nursery	LKTR	28.9962	29.0023	0.0061	0.0210
	ARGR	0.0000	0.0000	0.0000	0.0000
	RNWH	12.6974	12.6998	0.0024	0.0187
	LNSC	0.0000	0.0000	0.0000	0.0000
	BURB	17.1992	17.2035	0.0043	0.0250
	SLSC	22.3866	22.3839	-0.0026	-0.0118
	LKCH	96.7500	96.7617	0.0117	0.0121
Total		**1351.2754**	**1351.3012**	**0.0258**	**0.0353**[3]

1 HUs available in the pre-development phase represent the number of HUs present on shoals, shorelines, and in deep water areas of Snap Lake.

2 Closure based on the scheduled closure of the water management pond, which is associated with the mine water diffuser.

3 Sum of %change column.

Note: LKTR = lake trout; ARGR = Arctic grayling; RNWH = round whitefish; LNSC = longnose sucker; BURB = burbot; SLSC = slimy sculpin; LKCH = lake chub.

Table 9.5-29 Total Habitat Units Lost or Created in Snap Lake for Each Fish Species

Species	Total HUs Lost or Gained, Construction/Operations	Total HUs Lost or Gained, Post Closure
LKTR	0.0116	0.0179
ARGR	-0.0072	-0.0045
RNWH	-0.0114	-0.0075
LNSC	-0.0067	-0.0048
BURB	-0.0045	-0.0019
SLSC	-0.0102	-0.0080
LKCH	0.0300	0.0345
Total	**0.0015**	**0.0258**

[1] Total indicates a net gain in HUs after construction and mitigation.

Note: LKTR = lake trout; ARGR = Arctic grayling; RNWH = round whitefish; LNSC = longnose sucker; BURB = burbot; SLSC = slimy sculpin; LKCH = lake chub.

Table 9.5-30 Predicted Dust Accumulation Along the Perimeter of the Northwest Peninsula, Snap Lake

Map Position	Easting (UTM)	Northing (UTM)	TSP Deposition Rate [kg/m²/yr]	TSP Density [kg/m³]	Dustfall [mm/yr]
1	506531.0	7052196.0	0.005297	1220	0.004340
2	506910.2	7052517.4	0.009919	1220	0.008130
3	507003.4	7052960.9	0.009445	1220	0.007740
4	507109.1	7053378.4	0.004745	1220	0.003890
5	506674.8	7053425.9	0.006831	1220	0.005600
6	506252.2	7053191.3	0.010845	1220	0.008890
7	505789.3	7053026.8	0.004701	1220	0.003850
8	505411.9	7053083.7	0.002932	1220	0.002400
9	505131.3	7053361.3	0.001936	1220	0.001590
10	504781.9	7053103.6	0.001730	1220	0.001420
11	504334.7	7053070.5	0.001256	1220	0.001030
12	503904.3	7053117.4	0.000925	1220	0.000759
13	503426.4	7053084.2	0.000704	1220	0.000577
14	502949.0	7052937.3	0.000553	1220	0.000453
15	502489.8	7052853.0	0.000445	1220	0.000365
16	502571.4	7053078.9	0.000457	1220	0.000375
17	502266.5	7052793.0	0.000405	1220	0.000332
18	501854.7	7052661.1	0.000343	1220	0.000281

TSP = total suspended particulates.

Dust Deposition

Immediately adjacent to the mine, maximum annual dust deposition in Snap Lake is predicted to be 0.0088 mm/yr

Calculations using deposition rate of dust for various locations around Snap Lake were undertaken to determine potential dust accumulation over one year. Dust accumulation was determined for 18 locations around the northwest peninsula (Table 9.5-30, Figure 9.5-11). The average accumulation is predicted to be 0.00289 millimetres per year [mm/year], well below the 1 mm threshold for fish egg survival. Predicted accumulations immediately around the project ranged from 0.00889 to 0.00142 mm/yr and decreased steadily as the distance from the mine increased. At the extreme west end of the lake, accumulations of 0.000281 mm/yr were predicted.

Maximum dust accumulation over known lake trout spawning areas is predicted to be 0.0009 mm/yr

Accumulations over the active lake trout spawning beds mirror the northwest peninsula data. Dust accumulation was calculated for 11 spawning areas distributed throughout the eastern section of Snap Lake (Table 9.5-31, Figure 9.5-11). Average accumulation over the spawning areas was 0.000508 mm/yr with accumulation ranging from 0.000226 to 0.0009 mm/yr. The greatest accumulation was at the site closest to the project, 0.0009 mm/yr, and decreased as the distance from the mine site increased, with accumulations of 0.000226 mm/yr at the furthest spawning bed.

Table 9.5-31 Predicted Dust Accumulation Over Active Lake Trout Spawning Beds in Snap Lake

Spawning Bed Position	Easting (UTM)	Northing (UTM)	TSP deposition rate [kg/m²/yr]	TSP density [kg/m³]	Dustfall [mm/yr]
1	508471.9	7053355.2	0.000896	1220	0.000734
2	508055.4	7052735.3	0.001098	1220	0.000900
3	508609.1	7052573.3	0.000730	1220	0.000599
4	510096.2	7052330.2	0.000327	1220	0.000268
5	510521.8	7052132.9	0.000275	1220	0.000226
6	508916.0	7052138.1	0.000648	1220	0.000531
7	510275.7	7051539.2	0.000336	1220	0.000275
8	509239.5	7051190.6	0.000593	1220	0.000486
9	508117.1	7050788.9	0.000815	1220	0.000668
10	508843.4	7050329.9	0.000530	1220	0.000434
11	508363.9	7050113.7	0.000564	1220	0.000462

TSP = total suspended particulates.

Figure 9.5-11 Dust Deposition to Aquatic Habitats

The annual dust fall accumulations predicted for areas of Snap Lake fish habitat are <1 mm/ yr

These calculations illustrate that dust deposition will be minimal and mainly isolated to the mine site as accumulation drops off the further from the project a sampling site is located. The annual dust fall accumulation for Snap Lake of 0.00199 mm/yr is less than the 1 mm threshold limits for effect on fish and fish habitat on an annual basis. As a more conservative estimate, if dust deposition were to remain in place on any of the habitats identified, the total accumulation over the life of the project (26 years) would remain less than 1 mm. Along the northwest peninsula the greatest potential accumulation is 0.231 mm of material over the 26 year time frame. However, this is a worst case scenario as wind and wave action will continue to act in the lake to redistribute all fine materials to deeper water depositional areas. Wind and wave action are continual forces in lakes acting to clean rocky shoal and shoreline areas.

Change to Hydrology

Fish habitat in streams affected by the project footprint did not change as a result of hydrologic effects

Two sub-basin watersheds, S and Q, were identified during the baseline stream habitat assessments as having streams with the potential to provide fish habitat. These streams, S1(WQ) and S27(WQ), were both found to have fish present during the spring of 1999. Based on the sub-basin impact assessment presented in Section 9.3.2 for the hydrologic environment, the construction of mine infrastructure in these two watersheds will result in no change to drainage area (see Table 9.3-32) and, therefore, no change to surface water runoff. As a result, no changes to fish habitat as a consequence of effects to the volume of flows, persistence or duration of flows, and timing of flows are expected.

Changes in volumes of flow from outlet stream for the north lake is predicted to occur during the operations period and could affect fish habitat

Based on the reduction of groundwater flow to the north lake during mine operation, a reduction in outflow from this lake is predicted. This reduced outflow will also be carried forward as a reduced outflow for two small lakes and the northeast lake downstream (Section 9.3.2). For the outlet stream from the north lake, the maximum reduction in discharge is predicted to be approximately 8% at the end of the operations period. During most of the operations period, the reductions in discharge will be less than this (*i.e.*, a gradual increase in effect will occur as the underground operations expand to a maximum level). These reductions in outlet stream discharges will cease once groundwater flow returns to baseline conditions at the end of the operations period. In the Hydrology Impact Assessment (Section 9.3.2) the environmental consequence of these flow reductions was rated as low for the north lake.

The impact to fish habitat in the outlet streams from the north lake in the local study area may occur and has been assessed as having a low magnitude

Presently little is known about the nature of the outlet stream from the north lake. Because the reduction of flows will occur throughout the year, but in varying magnitude, there are several potential impacts. These include reduced spawning habitat, reduced rearing habitat, and reduced migratory capacity which may result in the stranding of juvenile and adult fish in areas incapable of supporting overwintering (*e.g.,* in NL5 or NL6, or areas of the stream channel). Based on the discussion presented in Section 9.3.2, it is likely that the greatest proportional change in outflow will occur during low flow periods (*i.e.*, during winter, summer or fall base flows) with relatively smaller reductions in flow during the high spring run-off period. As well, a reduction in flows by 8% would still be considered to be within the historical range of low flow events. However, the lack of baseline data for both the hydrology and fish habitat in these streams makes it difficult to exclude any potential impact, or fully assess the potential magnitude, based on the criteria defined in Section 9.1. As a conservative estimate, this impact has been evaluated as potentially having a low magnitude whereby up to 10% of the habitat may be affected.

No changes to fish habitat as a result of effects to sediment yield are expected

During construction and operation of the Snap Lake Diamond Project there will be instances where natural ground cover may be disturbed and maintenance activities may increase the possibility of extra sediment entering surface water. The possibility of increased sediment altering fish habitat within Snap Lake and local streams was identified. Based on the sub-basin impact assessment presented in Section 9.3.2 for the hydrologic environment, increased sediment yield in surface water will be negligible. A series of three seepage and runoff collection ponds will reduce suspended solids in the runoff originating from the north pile. All surface flow from the north pile will be diverted through these ponds. Runoff collected in ditches and through culverts may contain elevated solids. Due to the small drainage areas on the northwest peninsula, there is insufficient runoff to develop permanent open water channels. As a result, runoff will tend to pass through low-lying areas containing muskeg and small lakes at low flow rates which will cause most suspended sediment to be retained prior to reaching streams S1 (WQ), S27 (WQ), or Snap Lake.

Change to Non-fish Aquatic Organisms

The linkage between changes to non-fish aquatic organisms and fish habitat is valid for Snap Lake during operations but was evaluated as having a negligible magnitude

Based on the evaluation of potential effects to or on non-fish aquatic organisms in Snap Lake, there is a potential for calcium to have an influence on the zooplankton community structure. As discussed in Section 9.5.2.2, calcium is an essential nutrient but in excess it is known to cause reproductive effects (Biesinger and Christensen 1972). In general, Copepods and Daphnia are more susceptible to toxicological effects than other zooplankters, namely rotifers. Based on the spatial extent and the predicted concentration of calcium, the effect on zooplankton was rated conservatively as low. This conclusion was conservative, since maximum predicted concentrations are roughly equivalent, but do not exceed, the LOEC for sensitive cladoceran species. Furthermore, the average predicted calcium concentration (90 mg/L) would be well below the LOEC for *Daphnia magna*. Based on this, a substantial change to the fish food resource, in relation to biomass and composition, in Snap Lake is not expected. No change in fish habitat is predicted and therefore an impact magnitude rating of negligible has been assessed.

The linkage between changes to non-fish aquatic organisms and fish habitat is valid for the north lakes, post-closure

Based on the evaluation of potential effects to phytoplankton, zooplankton, and benthic invertebrates in the north lake, NL5, NL6, and the northeast lake, there are several possible influences on community structure for these organisms. The majority of these parameters of concern are related to the inflow of porewater to the north and northeast lakes over a substrate area of less than 10% in each lake during the post-closure period. The chemicals of concern with predicted environmental consequences greater than negligible include:

- Chromium (trivalent and hexavalent chromium in porewater for both the north and northeast lakes; trivalent chromium in the water column for the north lake, NL5 and NL6) is predicted to have a low environmental consequence and may result in slight reductions in biomass for phytoplankton, zooplankton, and benthic invertebrate. Some selection against more sensitive benthic organisms is also anticipated, but the dominant species expected to occur would be tolerant to predicted chromium concentrations.
- Copper in porewater of the north and northeast lakes is predicted to have a low environmental consequence and may result in a slight reduction in phytoplankton biomass and preferential community selection against diatoms. It may also result in a slight reduction in secondary productivity (both zooplankton and benthos) with a preference shift away from cladocera in favour of rotifers (more tolerant).
- The north and northeast lake sediments will receive pH-affected porewater in excess of the tolerance range for most phytoplankton, zooplankton, and benthic species (pH of 11.8 is predicted). In addition, an increase in pH may cause heightened ammonia concentrations. The pH will dissipate relatively quickly to overlying waters as there is little support for establishment of a pH gradient. A reduction in biomass and preferential selection for alkali species is expected. This impact was rated as having a moderate environmental consequence.

- Nitrate is predicted to have a moderate environmental consequence on phytoplankton in both the north and northeast lakes and a low environmental consequence on zooplankton and benthos. A combination of increased soluble nitrogen in the presence of algae capable of maximizing low phosphorus concentrations may lead to an increase in production. The current population is dominated by diatoms and will not change appreciably. The impact to food fish organisms will be a rise in phytoplankton biomass with a subsequent smaller increase in zooplankton and benthos production.
- Aluminum in the porewater of both the north and northeast lakes is predicted to have a low environmental consequence on phytoplankton, zooplankton, and benthic invertebrates. Aluminum will affect less than 10% of the lake sediments in each lake and, overall, reduced production across phytoplankton, zooplankton and benthic species is expected. Aluminum is known to bioconcentrate and a minimal accumulation through the food chain is also expected.

The primary changes predicted for non-fish aquatic organisms are related to reductions in productivity and possible shifts away from less tolerant species

In relation to a change in habitat quality for fish in the north and northeast lakes, the type of change expected and the spatial extent are both important to consider. Based on the evaluation completed in Key Question F1, the types of changes to non-fish aquatic organisms expected are primarily reductions in productivity. Some species selection toward more tolerant species may also occur, but is not predicted to affect any of the expected dominant species. The spatial extent of the impacts are limited to less than 10% of the area of each lake. There is also the possibility of an increase in productivity as a result of the increased nitrate concentrations. Since 90% of the substrate areas of these lakes will be unaffected, and reduced productivity or community structure shifts throughout the lake are unlikely. As a conservative estimate, the impact magnitude rating for these effects to fish habitat is rated as low.

9.5.2.3.4 Residual Impact Classification

Based on the impact analysis, Table 9.5-32 summarizes the residual impacts and presents the environmental consequences for the potential effects to fish habitat.

Construction and Removal of Instream Structures

Changes to fish habitat during both the construction/operations and post-closure periods were evaluated for magnitude of effect

Habitat in Snap Lake was categorized in relation to its suitability to fish in various life stages (*i.e.*, spawning, rearing, foraging, refuge). An ecological threshold, or level beyond which a specific use is affected, was used to assess the magnitude of a potential effect. The assessment of the effects during the construction and operations phase was based on the proposed maximum mine development. The residual affect assessment of the closure footprint was based on partial removal of the water intake structure and the removal of the waste water outflow pipe and diffuser.

Table 9.5-32 Residual Impact Classification for Changes to Fish Habitat

Waterbody	Activity	Direction	Magnitude	Geographic Extent	Duration	Reversibility	Frequency	Environmental Consequences
Snap Lake	dust deposition	negative	negligible	local	medium-term	reversible (short-term)	medium	negligible
	instream structures (construction/ operations)	positive	negligible	local	short-term	irreversible	low	n/a
	instream structures (post-closure)	positive	negligible	local	short-term	irreversible	low	n/a
	construction/ operations-sediment yield	negative	negligible	local	medium-term	reversible (short-term)	medium	negligible
	change to food source resulting in change to habitat	negative	negligible	local	medium-term	reversible (short-term)	high	negligible
Streams, Snap Lake basin	mine footprint	negative	negligible	local	short-term	reversible (short-term)	low	negligible
North lake	change to food source resulting in change to habitat	negative	low	local	long-term	reversible (long-term)	high	low
Northeast lake	change to food source resulting in change to habitat	negative	low	local	long-term	reversible (long-term)	high	low
Outlet stream, north lake	groundwater changes	negative	low	local	medium-term	reversible (short-term)	medium	low

Note: Method used to determine environmental consequence is explained in Section 9.1.

n/a = not applicable. The environmental consequences of positive impacts are not assessed.

A net gain of fish habitat is predicted during construction and operations of the water intake and waste water diffuser

The construction and operations of the water intake and mine water diffuser will result in the alteration or loss of fish habitat. This loss will be countered simultaneously during construction of the water intake and the mine water diffuser through mitigation. The structures will be constructed to create habitat for all fish species in Snap Lake. The habitat created will result in a net gain of 0.0015 HUs.

An additional gain of habitat is also predicted at closure when the diffuser and pipeline are removed from Snap Lake

At closure the diffuser structure and associated pipeline will be removed from Snap Lake. This will result in the temporarily altered habitat around these structures being returned to the available habitat in Snap Lake. The net gain of HUs at closure is 0.0258. This is based on 0.0160 ha of deep water being reclaimed, with the removal of the pipeline and diffuser, in addition to the 0.1568 ha of habitat created through mitigation during construction and operations.

Overall the habitat area affected by the construction of the water intake and the mine water diffuser is less than 1% of the aquatic habitat in Snap Lake; the environmental consequence is predicted to be negligible

The effect of the instream structures will be negligible for all fish species as there is an abundance of all habitat types for all fish species found throughout Snap Lake. There are no uncommon or unique habitat types found along the northwest peninsula that could be affected by the construction of the instream structures. Overall the habitat area affected by the construction of the water intake and the mine water diffuser is less than 1% of the aquatic habitat in Snap Lake. The effect will be confined to the local area and the duration of the effect will be short-term since it will occur during construction. The net gain in habitat is a positive impact that will remain after closure; therefore, it is an irreversible gain.

Dust Deposition

The environmental consequence of dust deposition is predicted to be negligible

Dust deposition on an annual basis is predicted to be below the 1 mm/yr threshold for an affect to fish egg survival. Based on this, the magnitude is predicted to be negligible. The geographic extent is limited to the LSA and the deposition will occur for the life of the mine so the duration is medium term. The effect is reversible as natural wind and wave action of the lake will continue to keep some areas of habitat free of fine sediments while accumulating these sediments in the depositional areas of the lake. The frequency is medium as dust deposition will occur intermittently on an annual basis during operations. Overall, the environmental consequence is negligible.

Change to Hydrology

The environmental consequences of changes to hydrology affecting fish habitat in the local study area are predicted to be negligible

For the LSA inland lakes and streams, the magnitude of a change to fish habitat, from a change to hydrology, was defined as negligible. No fish bearing inland lakes are found within the runoff containment system for the proposed project. For the two sub-basins with fish bearing streams and some infrastructure development, no changes to volume, timing, or duration of flow are predicted. Therefore the environmental consequence of hydrologic change for these waterbodies is negligible for the proposed project.

The environmental consequences of changes to hydrology affecting fish habitat in outlet streams from the north lake is predicted to be low

The changes in groundwater flow are expected to alter flows in outlet streams from the north lake. The potential impact to fish habitat in these streams was evaluated as having a low magnitude and low environmental consequence. The potential impact will occur during the operations period, with the maximum level of flow reduction occurring at the maximum underground development stage. The impact is reversible and will cease at, or shortly after, closure. In addition, the reductions in flows will likely fluctuate on a seasonal basis so the frequency was evaluated as medium. There is, however, a high level of uncertainty regarding the extent of habitat alteration as a result of the limited information available on hydrology and fish habitat in these lakes and streams.

Change to Non-fish Aquatic Organisms

The overall environmental consequence to the fish food source in the north lake and northeast lake is predicted to be low and negligible in Snap Lake

Changes to non-fish aquatic organisms through reductions in productivity and minor species selection toward more tolerant species are predicted as potentially occurring in the north and northeast lakes. The impact will occur post-closure and is related to primarily elevated metal concentrations and pH in groundwater inflows. The impact is reversible in the long-term since no species losses are expected and, once baseline chemistry conditions return in the porewater, productivity and community structure would be expected to return to baseline conditions over time. The magnitude of the impact to non-fish aquatic organisms, as a food source for fish (fish habitat), is rated as low. The spatial extent of the impact is limited to less than 10% of the area of each lake. With these factors in mind, the environmental consequence is rated as low. In Snap Lake, the environmental consequence of changes to non-fish aquatic organisms was rated as negligible when these organisms are considered as a food resource for fish.

9.5.2.3.5 Monitoring

Total suspended solids will be monitored when intake and outlet structures are constructed

A total suspended solids (TSS) monitoring program will be conducted during the construction of the water intake and diffuser structure in Snap Lake. This will provide a review of the effectiveness of sediment control efforts and ensure compliance with regulatory requirements.

Fish habitat use monitoring at the inlet and outlet structures is recommended

Fish habitat use around inlet and outlet structures in Snap Lake will be monitored after construction. This information can be used to recommend any changes that may be required at closure to improve the habitat provided by the rock-filled pads.

9.5.2.4 Key Question F-3: What Impacts Will the Snap Lake Diamond Project Have on Fish Health?

Potential acute and chronic effects on fish health were assessed for the application and post-closure cases

The potential acute and chronic effects of the Snap Lake Diamond Project on fish health were assessed. During construction and operations, fish have the potential to be exposed to changes in water and sediment quality within Snap Lake. These changes will result from direct exposure to the treated combined discharge, north pile and WMP seepage, and runoff from the site into Snap lake. During post-closure, fish health within the LSA may be compromised as a result of groundwater recharge to the north lake and northeast lake.

Three pathways of exposure were evaluated

Potential pathways of chemical exposure to fish are via direct exposure to water, direct exposure to sediment and indirect exposure through dietary uptake of food organisms. As part of the impact assessment on fish health, these three pathways were considered and reviewed with regard to potential linkage to fish health within the LSA.

9.5.2.4.1 Linkage Analysis

Three potential linkages were analyzed

The following potential linkages between the Snap Lake Diamond Project and acute or chronic health effects on fish were analyzed:

- linkage between changes in water quality and acute or chronic effects on fish health;
- linkage between changes in sediment quality and acute or chronic effects on fish health; and,
- linkage between changes to non-fish aquatic organisms and acute or chronic effects on fish health.

Changes in Water Quality

The linkage to acute effects is invalid; however, the linkage to chronic effects is valid

The impact assessment on water quality (Section 9.4.2) indicated the potential for low environmental consequences to the water quality within Snap Lake. The potential impacts are strictly related to chronic level effects to aquatic biota; therefore, the linkage to acute effects to fish is invalid. These conclusions are consistent with literature values and supported by the site-specific whole-effluent toxicity tests (Appendix IX.8) completed using site water collected during the AEP. In these tests, acute toxicity to rainbow trout and fathead minnow was not present for the tests using treated mine water.

Chromium and total dissolved solids were carried forward from the water quality assessment to the application case

The linkage between changes to water quality in Snap Lake and possible chronic effects on aquatic biota was carried forward for parameters predicted to cause greater than negligible impacts during construction and operations. The linkage is related to elevated water concentrations above the site-specific benchmark (HC_5) for hexavalent chromium. No other metals were carried forward from the water quality assessment because none exceeded the site-specific benchmarks for >1% of the waterbody (and therefore none exceeded negligible). In addition to chromium, TDS was carried forward from the water quality assessment because of the projected increase in TDS over baseline concentrations.

Chromium was carried forward from the water quality assessment of the post-closure case

The linkage between changes to water quality during post-closure and possible chronic effects on aquatic biota was carried forward for chromium in the north lake and in NL5 and NL6 downstream of the north lake. The concentrations of trivalent chromium exceeded the HC_{10} benchmark value in greater than 20% of the north lake (therefore, the residual impact was rated as moderate for water quality). Other parameters were evaluated as negligible and not considered for further assessment because concentrations did not exceed general guidelines or the site-specific HC_5 benchmark value in more than 1% of the lake volume or area (and therefore were rated as causing negligible impact). Water quality changes in the northeast lake water column were predicted to be negligible for all parameters. Therefore no further assessment was conducted.

Changes in Sediment Quality

The sediment linkage is minimal, but valid for operations

The treated combined discharge from the Snap Lake Diamond Project to Snap Lake will consist primarily of fine suspended sediment and dissolved metals and major ions. The dispersion and mixing of the mine water within the water column is high during the open water periods and results in low concentrations within Snap Lake. While the majority of the material will remain dissolved, biological uptake and subsequent deposition to the sediments will occur. However, deposition of metals to sediments is expected to be minimal except for chromium (Section 9.4).

Only chronic effects of chromium were valid linkages during operations

Based on the water quality conclusion of a low environmental consequence from elevated hexavalent chromium within Snap Lake, changes to water quality may lead to increased sediment chromium concentrations with possible chronic effects to aquatic biota. This pathway was thus considered valid and carried forward into the impact assessment. Concentrations of chromium in sediment are not expected to approach acutely toxic levels; therefore the linkage to acute effects is invalid.

The linkage between sediment quality and fish health is valid for the post-closure case

Groundwater recharge to the north lake and northeast lake during post-closure will contribute elevated metals via the sediments into both the north lake and the northeast lake. The aerial extent of the porewater area predicted to be effected is less than 10% for both lakes, but greater than 1%. Due to the process in which the parameters of concern enter into the north lakes, a conservative approach assuming equivalent sediment porewater concentrations to predicted groundwater was assumed in the water quality impact assessment. Based on these conservative assumptions, the potential linkage between a change to sediment quality and an effect on fish health is valid for both the north lake and northeast lake.

Changes in Non-fish Aquatic Organisms

The linkage between changes to non-fish aquatic organisms and fish health is valid for both operations and post-closure

Metals associated with the Snap Lake Diamond Project may accumulate from water/sediment to the tissues of aquatic organisms. Increased concentrations of metals in Snap Lake, the north lake, and the northeast lake may lead to increased uptake by phytoplankton, zooplankton, and benthic invertebrates. The increased metal concentrations in prey species may lead to increased uptake by fish through the food chain. Increased metal concentrations in fish may exceed tissue thresholds for effects on fish health. Therefore, the linkage between changes to non-fish aquatic organisms and potential chronic effects on fish is a valid pathway.

9.5.2.4.2 Mitigation

No additional mitigation measures beyond those already planned for water quality are indicated for the application case

Potential impacts of the Snap Lake Diamond Project on fish health are directly related to changes in water quality from the discharge of treated mine water, seepage, and surface drainage water. Mitigation plans have been discussed in the water quality assessment (Section 9.4) and the WMP (Appendix III.4). The measures discussed in those sections will substantially reduce potential impacts to fish health. No additional mitigation measures are indicated for construction and operations.

The post-closure case may produce impacts to fish health but uncertainty is high; therefore, mitigation measures will follow further studies

The post-closure case may produce impacts to fish health given the predicted water quality and sediment porewater quality in the north and northeast lakes. However, there is high uncertainty in the post-closure predictions. Therefore, mitigation measures cannot be recommended with any confidence. Follow-up investigations to reduce the uncertainty of the post-closure water quality predictions proposed and ongoing. Results of these studies will be reviewed before further mitigation planning can proceed.

9.5.2.4.3 Impact Analysis

The water quality assessment formed the basis for the fish health assessment

The water quality assessment provided an initial screening of predicted maximum concentrations compared to CCME guidelines, U.S. EPA criteria, or site-specific benchmarks. Of the parameters assessed, only chromium and major ions (*i.e.*, TDS) were identified as chemicals of concern in Snap Lake. These chemicals of concern were carried forward into the fish health assessment for construction and operations. Chromium was also carried forward for the assessment of water column impacts during post-closure for the north and northeast lakes. In addition, chromium, molybdenum, copper, and aluminum were carried forward for the assessment of impacts from changes in porewater quality in the north lakes, post-closure.

The magnitude criteria used in the assessment of impacts included consideration of exceedance of chronic effect benchmarks, as well as spatial and temporal extent and indirect effects

The magnitude criterion used in the assessment of impacts on fish health included consideration of exceedance of chronic effect benchmarks, as well as spatial and temporal extent of changes in water or sediment quality (Table 9.5-1). The definition of negligible, low, moderate and high magnitude impacts on fish health had to account for the fact that impacts to fish health would be a function of dose, not simply concentration. Dose is determined by chemical concentration, duration of exposure, and spatial extent of changes in chemical concentration. Indirect effects via impacts on fish food organisms were also considered (*e.g.*, reduction in growth because of a reduction in fish food organism abundance).

Comparing predicted concentrations with published chronic effect values was the first step in assessing magnitude of impacts from exposure to metals

The first step in assessing the magnitude of potential impacts of metals on fish health was to compare predicted concentrations with published chronic effect values for relevant fish species. The "chronic effect value" is the geometric mean of the LOEC and the NOEC (Suter and Tsao 1996). The chronic effect value data were evaluated for relevance to Snap Lake and the presumed north lake fish species using the general rules for using a subset of toxicological data (MacDonald *et al*. draft) (Appendix IX.8).

The potential magnitude of impacts of total dissolved solids on fish health was assessed by comparing predicted concentrations with published LOECs from laboratory studies and with observational data from the field

The potential impacts of TDS on fish health in Snap Lake were assessed by comparing predicted concentrations with published LOECs from laboratory studies and with observational data from the field. There are almost no laboratory studies of TDS; rather, laboratory tests are usually done on individual ions such as chloride. Therefore, LOECs for the ions contributing the largest portion to TDS were examined. The dominant ions are calcium and chloride. Observational data from the field usually refer to TDS; however the constituents of the TDS vary from sodium chloride to sodium/magnesium to sodium bicarbonate lakes. Observational data of the effects of road salting were also useful. However, these data focussed on chloride effects rather than the cations associated with chloride.

If predicted concentrations exceeded chronic effect values, then the spatial and temporal extent of exceedances was examined

The magnitude of impacts on fish health depends not only on exceedance of chronic effect values, but also on the area or volume of the lake affected by these exceedances, and the length of time over which the exceedances occur. The area or volume of the lake affected is important because fish have behavioural responses to changes in water quality and avoid areas with elevated metals, including chromium (Beitinger and Freeman 1983; Atchison *et al*. 1987). Therefore, if a small area or volume is affected by increased metal concentrations, there would still be large areas for fish to move to. Similarly, a small area or volume affected by increased TDS would leave large areas where fish would not be affected. The predicted changes in water quality are confined to deep basin areas (below 8 m depth) in winter (Section 9.4.2). These deep basin areas were compared to critical and preferred fish habitat in order to ensure that the evaluation of spatial extent included consideration of particular areas where fish may congregate. The length of time that metals or TDS are expected to be elevated is also important to the estimation of magnitude. Elevated metal or TDS concentrations over one season would be expected to have a lower magnitude of impact than elevated concentrations persisting year-round. The season when increased concentrations occur is also important; increased concentrations during periods of early life stage development would have a higher potential magnitude than other seasons.

Indirect effects were included in the magnitude evaluation

The magnitude of indirect effects was evaluated by determining the potential for exposure of fish via the food chain and by evaluating effects on fish health via effects on the abundance of fish food organisms.

Other impact assessment criteria were identical to those used in the water quality assessment

The other impact assessment criteria were identical to those used in the water quality assessment (Table 9.1-3). Direction was either neutral (no change in fish health) or negative (a decrease in fish health). Geographic extent was either local (restricted to Snap Lake or the post-closure north lake) or regional (the RSA). Duration was either short-term (pre-construction and construction), medium-term (the 26 years of operation) or long-term (26+ years: extending into post-closure). Reversibility was short term (reversible within 30 years), long-term (reversible in greater than 30 years) or irreversible. Frequency was low (occurs once), medium (occurs intermittently) of high (occurs continuously).

Snap Lake: Construction and Operations

Chromium

Direct Effects

Direct exposure to chromium is the primary pathway of concern

As indicated in the linkage analysis, the pathway of primary concern to fish health is direct exposure to elevated hexavalent chromium concentrations via water. Hexavalent chromium was the only parameter that exceeded either the general CCME guideline or site-specific benchmark within greater than 1% of the lake volume or area.

Predicted maximum chromium concentrations are less than the chronic effect value for relevant salmonid species

Assuming that all chromium released from the treated combined discharge is in the hexavalent form, the maximum hexavalent chromium concentration in the discharge is predicted to be 7.5 μg/L. The maximum water column concentration of hexavalent chromium at approximately 230 m from the discharge is estimated to be approximately 2.5 μg/L. Both the maximum discharge concentration and maximum concentration in lake water are less than the chronic effect value for relevant salmonid species (73.2 μg/L for rainbow trout). Therefore, no direct toxic effects on Snap Lake fish are expected, since even the most sensitive members of the salmon family (including lake trout and whitefish) would not be affected.

Since no direct toxic effects are expected, no further evaluation of spatial or temporal extent is required

Further assessment of the magnitude of direct impacts from increased hexavalent chromium concentrations in the water column is not required since concentrations are not predicted to exceed the chronic effect value even in 100% discharge water before any dilution by lake water.

The magnitude of effects of chromium on fish health will be negligible

Direct toxic effects of predicted chromium concentrations in Snap Lake are not expected; therefore, the magnitude of impacts is rated as negligible.

Direct exposure via sediments was not evaluated quantitatively

Direct exposure to chromium in sediments was not evaluated quantitatively because sediment concentrations were not part of the predictive water quality model. Therefore, the assessment of the potential for impacts on fish health from chromium in sediment was based on professional judgement.

Chromium may be slightly increased in the sediments of Snap Lake

It is expected that the small increases in hexavalent chromium in the water column will produce small increases in sediment concentrations through biological uptake and subsequent deposition to the sediments. The magnitude of this increase in sediment chromium concentration cannot be estimated quantitatively; however, it is likely to be very low.

It is expected that the chromium in sediment will be unavailable to biota

Chromium in sediments is likely to be unavailable for uptake by aquatic organisms. It is well known that total metal concentrations in sediments do not correlate to biological effects (Chapman *et al*. 1998). Metal binding phases in sediments such as organic carbon, sulphides, and iron/manganese oxides can greatly reduce the bioavailability of metals. In addition, complexation of metals by ligands such as OH^- , Cl^-, HCO^{3-}, SO_4^{2-} and CO_3^{2-} has the potential to reduce bioavailability in porewaters (Chapman *et al*. 1998). Given the likely presence of metal-binding phases in Snap Lake sediments, the increase in Cl^- ligands because of the discharge of treated mine water, and the small increase in total chromium expected as a result of increases in the water column, exposure to chromium in sediments is unlikely to affect fish health.

Sediment concentrations will increase primarily in the area of Snap Lake affected during the winter

Water quality predictions indicate that the treated combined discharge will settle primarily within the deeper areas of Snap Lake during the winter as a result of density differences between the treated combined discharge and lake water. Under ice-covered conditions, the water quality near the sediments will be reduced; however, because of the relatively low concentrations and the metal-binding processes in the aerobic layer of the sediment (*i.e.*, iron and manganese oxyhydroxides and particulate organic carbon, Tessier *et al*., 1993, 1996) and acid-volatile sulphides in anaerobic sediments, the potential for direct effects to fish is not likely. In addition, because this occurs only in the deepest areas of Snap Lake, it is not expected to provide an exposure route to the early life stages of fish (eggs and larvae), which are usually the most sensitive life stages. Based on water concentrations predicted to only slightly exceed the general CCME guideline or benchmark value and bioavailability-limiting processes within the sediment, the impacts from increases in chromium in sediment are expected to be negligible.

Direct toxic effects of chromium in sediments on fish health are expected to be negligible

Direct toxic effects of chromium in sediments on fish health are expected to be negligible. This conclusion is based upon the likelihood of low concentrations combined with low bioavailability plus the fact that increased sediment concentrations will not correspond with locations in Snap Lake used by the more sensitive early life stages of fish (*e.g.*, spawning shoals and shoreline habitats identified in Figure 9.5-6).

Indirect Effects

The indirect exposure pathway for fish is through consumption of contaminated food

Indirect pathways of exposure are via fish food organisms and the uptake of chromium by these organisms. Site-specific Bio-Concentration Factors (BCFs) of between 150 and 165 for fish muscle tissue, and 150 and 380 for liver tissue, depending on species, were calculated for chromium from the Snap Lake baseline data (Appendix IX.11, Tables IX.11-9 and IX.11-10). By applying these values to the application case average mine water discharge concentration of 0.0078 µg/L for chromium, predicted tissue concentrations for chromium were calculated. Using this approach, values of 1.1 micrograms per gram (µg/g) for lake trout both in the liver and the muscle were predicted. Round whitefish tissue concentrations of 2.8 µg/g for the liver and 1.2 µg/g in the muscle are also predicted. Based on this, the predicted tissue concentrations for invertebrates and fish are very low. Therefore it is unlikely that accumulation of chromium in fish tissue via direct uptake or food chain transfer would result in tissue concentrations sufficient to cause health effects on fish.

Indirect effects via reduction in fish food organisms were also evaluated

Indirect effects on fish health via a reduction in the abundance of fish food organisms were evaluated by examining the potential for direct toxicity to these organisms. The maximum water quality concentration of chromium 230 m from the discharge in Snap Lake is expected to be 2.5 µg/L. This concentration is less than the chronic effect values for the three most sensitive species reported in the literature. These three species are all Cladocerans: *Daphnia magna* (3.32 µg/L), *Daphnia pulex* (6.13 µg/L) and *Simocephalus vetulus* (6.13 µg/L). The derived HC_5 for Snap Lake is 2.1 µg/L. Therefore, the maximum predicted water concentration 230 m from the discharge would slightly exceed the derived concentration estimated to cause sub-lethal effects to 5% of the aquatic community.

Indirect effects on fish health via food chain uptake of chromium or reduction in fish food organisms caused by chromium are expected to be negligible

Indirect effects on fish health via food chain uptake of chromium or reduction in fish food organisms caused by chromium are expected to be negligible. The very low BCFs (Appendix IX.11, Tables IX.11-9 and IX.11-10) for chromium would lead to negligible effects via food chain uptake. The slight exceedance of the HC^5 by maximum concentrations within 230 m of the discharge would potentially reduce the abundance of sensitive fish food organisms in a small portion of the lake (much less than 10%). The combination of the potential for sublethal effects on fish food organisms with less than 10% spatial extent of effects produces a negligible magnitude rating.

TDS Concentrations and Impact on Fish Health

Direct Effects

Predicted increases in TDS will not exceed the benchmark for defining a saline lake

Predicted TDS concentrations will be considerably higher than the baseline concentrations, but will be much less than the generally-accepted benchmark for the definition of a saline lake. The average baseline concentration for TDS in Snap Lake is very low at 15 mg/L. The proposed Snap Lake Diamond Mine is expected to raise the TDS concentration to a whole lake maximum of 330 mg/L (year twenty) in 10-20% of the lake area and to 444 mg/L in less than 1% of the lake. The primary toxicological concern of TDS is an increase in osmotic stress on aquatic biota. There are no TDS water quality criteria, but an accepted standard for classification of a saline lake is 3,000 mg/L (Hammer *et al*. 1975, Timms 1986, Mandaville 2002). At TDS concentrations greater than 3,000 mg/L freshwater biota begin to disappear (Hammer *et al*. 1975).

There is a wide range of tolerance to salinity among different fish taxonomic groups

Fish populations have been found to survive and reproduce in lakes with varying compositions and concentrations of major ions (Atton 1986, Hammer 1986). The sticklebacks (Gasterosteidae) and various members of the Salmonidae are able to withstand high levels of salinity, presumably due to their recent marine history. Members of the Percidae and Catostomidae families are less tolerant of prolonged exposure to high levels of salinity (Hammer 1986, Schryer 1994). A number of members of the Cyprinidae family are also tolerant, although their tolerance can vary considerably between species.

Reproductive impairment is the most common effect of chronic exposure to high salinity

The most common and first effect of prolonged exposure to high salinity levels is reproductive impairment (Rawson and Moore 1944, Burnham and Peterka 1975, Schryer 1994). The lowest observable concentration of salinity for which reproductive impairment has been observed in fish was 6,000 mg/L (Atton 1986) in northern pike (*Esox lucius*). Schryer (1994) observed that reproductive impairment in an introduced walleye (*Stizostedion vitreum*) population in a saline lake where the major ions were sodium and sulphate began to occur at concentrations of approximately 1,000 μ/S.

Effects on fish reproduction are not expected

The predicted concentration of TDS in Snap Lake is much lower than the LOEC for reproductive impairment in fish (330 mg/L compared to 6,000 mg/L). Furthermore, the maximum predicted concentration would be present in <1% of the lake. Therefore, effects on fish reproduction are not predicted in the application case.

Indirect Effects

Indirect effects will depend upon the exceedances of calcium and chloride toxicity thresholds for fish food organisms

Indirect effects from TDS will depend upon the potential for exceedance of calcium or chloride toxicity thresholds for fish food organisms. Chloride chronic effects begin at concentrations greater than 372 mg/L. The most sensitive calcium threshold is for cladocerans, at 116 mg/L (see Section 9.5.2.2).

Negligible indirect effects from chloride are predicted

Indirect effects of increased chloride concentrations via effects on fish food organisms are expected to be negligible. Maximum predicted chloride concentrations will not approach LOECs reported in the literature, even for the most sensitive species.

The magnitude of calcium effects on sensitive fish food organisms is expected to be low

Based on information presented in Section 9.5.2.2,, calcium effects on fish food organisms are expected to be low. Although there may be some chronic impacts from calcium on sensitive fish food organisms, these impacts are not expected to be sufficient to cause effects on fish health. The threshold for chronic impacts would only be approached during peak concentrations, would be limited to the winter months, and would be limited to 10-20% of the lake area. Therefore, it is highly unlikely that there would be a significant overall reduction in fish food abundance.

North Lake, Northeast Lake, NL5 and NL6: Post Closure

Increases in chromium, copper, molybdenum, nitrate, and pH are expected during post-closure

During post-closure, groundwater recharge from Snap Lake is predicted to flow into the north lake and the northeast lake within the LSA. Flow from the north lake will affect the downstream lakes NL5 and NL6. Increases in the following constituents are predicted:

- mixed-water column: trivalent chromium in the north lake and in NL5 and NL6;
- porewater/sediment: aluminum, trivalent and hexavalent chromium, copper, and molybdenum in both the north and northeast lakes; and,
- porewater/sediment: pH and nitrates in both the north and northeast lakes.

Chromium

Direct Effects

Chromium was carried forward because of predicted moderate impacts on water quality in the north lake and NL5 and NL6

Predicted water column concentrations in the north lake at post-closure suggest that exposure of fish to elevated concentrations of trivalent chromium may occur and will extend downstream into NL5 and NL6. As indicated in Section 9.4, the predicted maximum average water column concentration of trivalent chromium would fall between site-specific HC_{10} and HC_{20} site-specific water quality benchmarks in the entire north lake, as well as NL5 and NL6. Consequently, the magnitude of impact on water quality was rated as moderate in the north lake, NL5, and NL6 (Section 9.4.2).

Direct effects on fish health in the north lake and NL5 and NL6 from elevated trivalent chromium

Further assessment of the water column concentrations in the north lake indicates that the maximum predicted water column concentration of trivalent chromium within the entire north lake, and the two downstream small lakes, is approximately 12.3 µg/L. Water hardness in the north lake will be considerably lower than in Snap Lake, averaging less than 20 mg/L. Therefore, benchmarks for toxicity are lower than for Snap Lake. However, on comparing the literature values for a salmonid fish species, rainbow trout (*Oncorhynchus mykiss*) had a chronic value of 32.4 µg/L at a hardness of 20 mg/L, which is above the predicted maximum trivalent chromium value of 12.3 µg/L. Coho salmon (*Oncorhynchus kisutch*) was found to have a chronic effect value of 209.17 µg/L for trivalent chromium, at a hardness of 20 mg/L. Based on this, the impact magnitude related to trivalent chromium in the water column is negligible for the north lake, NL5, and NL6.

Elevated trivalent chromium concentrations in sediment porewater may have a negligible magnitude effect on early life stage fish

The predicted trivalent chronic effect concentration for chromium for fish (rainbow trout) is 32.40 µg/L, which is higher than the predicted porewater concentrations for trivalent chromium of 12.5 µg/L in the north lake and 4.5 µg/L in the northeast lake. Thus the impact due to trivalent chromium in the porewater on early life stages would be negligible for both lakes.

Elevated hexavalent chromium concentrations in porewater are predicted to have a low magnitude impact on early life stage fish

Elevated hexavalent chromium concentrations in sediment porewater of the north lake have the potential to affect early life stage fish. The porewater concentration of hexavalent chromium is predicted to be 313 µg/L in the north and northeast lakes. These concentrations are higher than the site-specific HC_{20} benchmark concentration, but are present only in the porewater. The affected porewater would occupy less than 10% of both the north lake and northeast lake. Based on the spatial extent and the site specific benchmarks, the impact was rated as having a low magnitude for both lakes. The effects on porewater will be continuous and have the potential to result in chronic effects to early life stage fish incubating or inhabiting the sediment-water interface area adjacent to the affected porewater.

Indirect Effects

The magnitude of impacts on benthic fish food organisms from increased trivalent chromium in porewater is predicted to be low in the north lake only

The maximum average chromium concentration in the majority of the north lake water column and sediments is above the HC_{10} (at hardness 20 mg/L) benchmark. The northeast lake porewater and water column concentrations are below the HC_5 benchmarks. The magnitude of impact to the north lake, zooplankton and benthos is low. This is based on both water column and porewater effects. A slight reduction in zooplankton and benthic production is predicted. The dominant chironomid taxa will not be affected and no substantial bioaccumulation is expected. Effects on porewater will be year-round rather than seasonal and represents a potentially chronic effect. The benthic organisms are all more tolerant to trivalent chromium and no lethal thresholds are exceeded. The magnitude for the northeast lake is rated as negligible.

The predicted magnitude of impacts on benthic organisms from elevated hexavalent chromium in porewater is low

Benthic organisms with toxicity thresholds below the predicted 313 µg/L on the Species Sensitivity Distribution for Hexavalent Chromium (Appendix IX.8, Table IX.8-6) include *Gammarus pseudolimnaeus* (11.33 µg/L), *Hyalella azteca* (106.42 µg/L), *Gammarus fossarum* (190.28 µg/L), and *Heptagenia sulphurea* (225.43 µg/L). *Hyalella azteca* was noted in baseline data from five reference stations and had a sample population greater than one in only one station (five members in WQR 7-3 in the Reference Lake). The dominant benthic organisms were midge larvae (Diptera; Chronomidae: Tantarsini, Chronomini). The chronic effect values, for hexavalent chromium, for these tribes of midge larvae were 9,679 µg/L and 10,304 µg/L respectively. The predicted concentration of hexavalent chromium does not approach the chronic value for the dominant organisms in the north and northeast lakes. However, it does exceed the benchmark for effects on *Hyallela* (a species that was present but not abundant in baseline samples from other nearby lakes). The spatial extent of the exceedance of the chronic effect value for *Hyalella* would be less than 10% of either lake and effects are year-round. The resulting combination produces a low impact magnitude classification for the north and northeast lakes.

The impacts on fish food organisms are predicted to have a low impact on fish health

Low impacts on fish food organisms from elevated trivalent and hexavalent chromium in porewater are expected to translate into overall low impacts on fish health. This conclusion is based upon the expectation that effects on fish food organisms will primarily be sublethal; therefore, total abundance of food organisms is not expected to decrease to the same extent as would be the case if lethal thresholds were exceeded. Furthermore, impacts on fish food species are expected in less than 10% of the north and northeast lakes. Therefore, predicted indirect impacts on fish health are low.

Copper

Direct Effects

Predicted copper concentrations in the water column and porewater do not exceed chronic effect values for relevant fish species

The predicted maximum water column concentration for copper is 1.1 µg/L in the north lake and 1.0 µg/L in the northeast lake during post-closure. These predicted copper concentrations in the water column and porewater do not exceed chronic effect values for salmonid fish species. The toxicity information presented in Appendix IX.8 (Tables IX.8-1 and IX.8-2) shows that the lowest measured chronic value reported for salmonids was for cutthroat trout (*Salmo clarki*) at 8.49 µg/L. The predicted concentrations in both water column and porewater are less than this chronic effect value for both lakes. Therefore, even the sensitive salmonid species in the north and northeast lakes would not be expected to be affected by copper during post-closure.

Copper in the water column and porewater predicted to have a negligible impact

Increased copper concentrations in the water column and porewater of the north and northeast lakes are predicted to have a negligible direct impact on fish health. This is because predicted copper concentrations are lower than the chronic effect value for sensitive salmonids.

Indirect Effects

The estimated magnitude of indirect effects on fish health via food chain transfer of copper and accumulation in fish tissues is low

Indirect effects on fish health may occur via uptake of copper into fish food organisms and subsequent transfer to fish tissues. Site-specific BCFs of between 1,813 and 3,349 for fish muscle tissue, and 19,100 and 53,947 for liver tissue, depending on species, were calculated for copper from the Snap Lake baseline data (Appendix IX.11, Tables IX.11-9 and IX.11-10). By applying these values to the post-closure case water column concentration of

1.1 µg/L in the north lake, predicted tissue concentrations for copper were calculated. Using this approach, values of 3,684 µg/g for lake trout muscle tissue and 59,342 µg/g in liver tissue were estimated. Round whitefish tissue concentrations of 1,994 µg/g for muscle tissue and 21,010 µg/g in liver tissue were also estimated. The LOEC for copper in fish tissue are 72 µg/g in the liver and 0.5 µg/g in the muscle of rainbow trout (Jarvinen and Ankley 1999). These tissue residues caused a 63% reduction in survival. However, the potential for elevated uptake into fish tissue will only occur in up to 10% of the area of the lakes. Therefore, impact magnitude is rated as low.

Indirect effects on fish health via effects on planktonic fish food organisms are also predicted to be low

The toxicity thresholds for key planktonic species will be exceeded in porewater but not the water column (Section 9.5.2.2). A reduction in phytoplankton and zooplankton productivity may affect fish health. The magnitude of the impact is rated as low because the spatial extent of decreases in productivity is predicted to be less than 10% of lake area.

Indirect effects on fish health from copper toxicity to benthic invertebrates in the north lakes are predicted to be low

Chronic effect values for most key benthic organisms of interest in the north lakes are well above the predicted porewater copper concentrations of 5.1 µg/L. For example, the chronic value for the worm *Nais* is 27.27 µg/L. The chronic value for the midges *Chironomus decorus* and *Chironomus tentans* are 252.59 µg/L and 466.78 µg/L, respectively. However, the concentrations will exceed the effect level for the snail *Physa* (4.5 µg/L). Since the predicted porewater copper concentration is below the chronic effect values for most but not all relevant benthic invertebrate species, low impacts are predicted on benthic invertebrates and, in turn, on fish health in both lakes. The low rating is based on a predicted spatial extent of less than 10% of the lake area.

Molybdenum

Molybdenum concentrations in the water column do not exceed guidelines but porewater concentrations do

Molybdenum water column concentrations in the north lake (3.3 µg/L) and northeast lake (1.3 µg/L) are expected to be below the general CCME aquatic life guideline of 73 µg/L. The porewater concentration in the north and northeast lake are expected to be above the CCME guideline at 81.1 µg/L.

Direct Effects

Molybdenum toxicity to fish is low and direct impacts on fish health are expected to be negligible

Recent studies of Pickard *et al.* 1997 and McDevitt *et al.* (1999) reported no toxicity to cutthroat trout embryo/alevin following a 30-day exposure to 90 mg/L molybdenum. Similarly, McDevitt *et al.* (1999) reported no toxicity to rainbow trout embryo/alevin or fry stages following a 60-day exposure to 30 mg/L molybdenum. These results indicate that the concentrations in the north lake and northeast lake porewater will have no effect on fish health and therefore the impact magnitude is negligible.

Indirect Effects

Predicted indirect effects of increased molybdenum on fish health are expected to be negligible

The magnitude of molybdenum toxicity to fish food organisms is predicted to be negligible (Section 9.5.2.2). Therefore, negligible effects on fish health are predicted via indirect effects on fish food organisms.

pH

The pH in the water column will be within CCME guidelines but porewater pH will exceed guidelines

The pH in the water column of the north and northeast lakes is predicted to be between 6.7 to 7.1 which is within the CCME guidelines limit of 6.5-9. However, in the porewater, the pH is predicted to be 11.8.

The expected post-closure pH of sediment porewater will affect the speciation of ammonia, metals and salts

The expected post-closure pH of 11.8 in the north and northeast lakes is highly alkaline. High pH values tend to facilitate the solubilization of ammonia, heavy metals, and salts. The precipitation of carbonate salts (marl) is encouraged when pH levels are high. An increase in pH may cause heightened ammonia concentrations (U.S. EPA 1986). Above a pH of 9, un-ionized ammonia is the predominant species (Morgan and Stumm 1981). Nitrate is expected to be elevated in porewater; this nitrate can be converted by sediment micro-organisms to ammonia. Under the elevated pH conditions, this ammonia will predominantly be in the more toxic un-ionized form (NRC 1997; Morgan and Stumm 1981). However, high pH also reduces the toxicity of certain metals like aluminum by binding them further. Higher pH is known to decrease cation mobility and increase anion mobility. The mobility of metals as hydrated ionic salts is dependent first upon which metallic element is participating as the positively charged ion (cation) and secondly, which anion makes up the negatively charged component of the salt (USGS 2002).

Direct Effects

The predicted porewater pH is in the lethal range for some aquatic organisms, including fish

Lethal effects of pH on aquatic life occur below pH 4.5 and above pH 9.5. The predicted porewater pH of 11.8 is expected to be lethal to most fish food organisms (Section 9.5.2.2) and early life stage fish.

Indirect Effects

Indirect effects of elevated pH are predicted because of effects on fish food organisms

Fish food organisms will be directly affected by high pH, as well as by elevated un-ionized ammonia concentrations created by the high pH conditions. These conditions will be lethal, reducing the abundance of food organisms in 1-10% of the lake area.

The magnitude of impacts from elevated pH on fish health is predicted to be moderate

The predicted pH in sediment porewater will exceed effects thresholds and will also lead to indirect toxicity via effects on ammonia speciation. These effects will be limited to less than 10% of the area in each of the north lakes. This spatial extent would place the pH effects in the low magnitude category for both lakes. However, effects will be year-round, and acutely lethal effects may occur (pH is the only stressor expected to cause acutely lethal effects). Therefore, as a conservative prediction, the impact magnitude of pH effects on fish health is rated as moderate.

Aluminum

Aluminum water column concentrations will be below CCME guidelines but porewater concentrations will exceed guidelines

Aluminum water column concentrations in the north lake (40 µg/L) and northeast lake (29 µg/L) are expected to be below general CCME aquatic life guidelines of 100 µg/L. The porewater concentration in the north and northeast lake are expected to be 468 µg/L, exceeding the aquatic life guidelines.

Aluminum is generally more toxic in acidic waters

Aluminum is generally more toxic in acidic waters over the pH range 4.4-5.4, with maximum toxicity occurring around pH 5.0-5.2, especially when the aluminum is in the supersaturated form (Schofield and Trojnar 1980). Even very low concentrations of aluminum (0.05 mg/L) can be lethal to white sucker fry (*Catostomus commersoni*) at pH 5.0 (Driscoll *et al*. 1980). Early developmental stages of fish were also found to be susceptible (Gun and Keller 1984). At this pH, essentially all aluminum is present as $Al(OH)_4^-$ and very limited $Al(OH)_3$ (Cooke *et al*. 1993; Morel and Hering 1993).

Direct Effects

The magnitude of direct effects on fish health from porewater aluminum concentrations is predicted to be low

At the predicted pH conditions, the highly toxic aluminate $Al(OH)_4^-$) ion will dominate. Exceedance of toxic thresholds will occur in less than 10% of the north and northeast lakes. On balance, the magnitude of direct effects is expected to below.

Indirect Effects

Indirect effects on fish health via effects on fish food organisms are expected to be low in both the water column and the porewater

Water column concentrations of aluminum in the north and northeast lakes are predicted to be less than the CCME guidelines; therefore, effects on planktonic fish food organisms are not expected. Low magnitude effects on benthic fish food organisms from elevated aluminum will occur in less than 10% of the north and northeast lakes.

Nitrate

Direct Effects

Nitrate is a nutrient that can be toxic to fish at relatively high concentrations

Nitrate is a major nutrient for aquatic plants. Nitrate can also be toxic to fish; the 96 hr LC_{50} values for chinook salmon (*Oncorhynchus tshawytscha)* and rainbow trout were 5.8 and 6.0 g/L, respectively (CCREM 1987). Steelhead and rainbow trout eggs and chinook salmon fry showed a increase in mortality at nitrate exposures of 5, 10, and 20 mg/L (CCREM 1987). There is no CCME numeric guideline for nitrate.

Toxic effects on fish health from the predicted nitrate concentrations may occur

The predicted nitrate concentration in the north and northeast porewater is 42.7 mg/L. This concentration exceeds some of the reported effects thresholds for early life stage fish. As discussed previously, this elevated concentration of nitrate is also predicted to increase the potential for ammonia toxicity in affected areas of sediment. Therefore, there is a potential for chronic effects on eggs and larvae via exposure to porewater concentrations of nitrate. Furthermore, nitrate may be transformed to ammonia and at the predicted pH, highly toxic un-ionized ammonia will dominate. This would create acutely lethal conditions for fish.

The magnitude of impacts from nitrate toxicity to fish is predicted to be moderate

The magnitude of impacts from nitrate toxicity to fish is predicted to be moderate for both lakes. The spatial extent of effects will be less than 10% of each lake. However, it is unknown whether any critical habitat areas (especially for spawning and rearing) will be affected. Furthermore, increases will be year-round and potentially lethal. Because of the uncertainty regarding critical habitat and the year-round nature of potential effects, the overall rating for impact magnitude is estimated as moderate.

Indirect Effects

Nitrate is a nutrient and could cause increases in the abundance of fish food organisms

Nitrate is a nutrient and could cause a moderate increase in phytoplankton biomass and a low increase in zooplankton and benthic biomass (Section 9.5.2.2). These increases are predicted assuming that the lakes are N:P co-limited. If such increases occur, the supply of fish food organisms would increase.

The tendency for nitrate to increase abundance of fish food organisms may be counter-balanced by ammonia toxicity

Although there may be stimulatory effects from increased nitrate concentrations in the sediments, these effects may be off-set by toxicity from ammonia created by microbial action on the nitrates combined with high pH. The net effect on fish food organisms cannot be predicted with any confidence.

9.5.2.4.4 Residual Impact Classification

Snap Lake Construction and Operations

The residual impacts on the fish health for Snap Lake are summarized in Table 9.5-33

All potential consequences on fish health from changes in water or sediment quality in Snap Lake are rated as negligible (Table 9.5-33). These ratings are driven by the impact magnitude ratings. The magnitude of direct effects on fish health from chromium or TDS was negligible because chronic values for effects on sensitive fish species were not exceeded by predicted concentrations. The magnitude of indirect effects on fish health from chromium or TDS were rated negligible because effects on fish food organisms were not predicted to exceed the chromium HC_5 benchmark for effects on the aquatic community. In addition, TDS concentrations are predicted to approach, but not exceed, LOEC concentrations for sensitive fish food species. Acute effects linkages were invalid because predicted maximum concentrations were always well below acute effects thresholds.

Table 9.5-33 Residual Impact Classification for Application Case Impacts of the Snap Lake Diamond Project on Chronic Effects on Fish Health in Snap Lake

Chemical of Concern	Direction	Magnitude	Geographic Extent	Duration	Frequency	Reversibility	Environmental Consequence
Hexavalent chromium in water column	negative	negligible	local	medium-term	high	reversible (long-term)	low
Hexavalent chromium in sediment	negative	negligible	local	medium-term	high	reversible (long-term)	low
Indirect effects of chromium via fish food organisms	negative	negligible	local	medium-term	high	reversible (long-term)	low
Increased TDS	negative	negligible	local	medium-term	high	reversible (long-term)	low
Indirect effects of TDS Via fish food organisms	negative	negligible	local	medium-term	high	reversible (long-term)	low

Note: Numerical score for ranking of environmental consequence is explained in Table 9.5.1.

The magnitude rating of negligible for all potential impacts results in a negligible environmental consequence to fish health in Snap Lake

Although any potential impacts to fish health in Snap Lake as a direct or indirect result of changes to water chemistry would be long-term, they would be reversible. The environmental consequence for all pathways has been rated as negligible for Snap Lake.

Snap Lake: Post-closure

No residual impacts are predicted for the post-closure period for Snap Lake

All discharges to Snap Lake will cease in the post-closure period and the water quality will gradually return to baseline conditions (see Section 9.4). Negligible impacts were predicted from elevated chromium in the application case; therefore impacts in post-closure would also be negligible before completely disappearing once chromium concentrations return to baseline.

North Lake, Northeast Lake, NL5 and NL6: Post-closure Case

The environmental consequence of potential impacts to fish health in NL5 and NL6 is low

The potential effects of chromium in NL5 and NL6 were rated as having a negligible environmental consequence (Table 9.5-34).

Table 9.5-34 Residual Impact Classification for Impacts of post-closure Groundwater Discharge to Fish Health in NL5 and NL6

Chemicals of Concern	Direction	Magnitude	Geographic Extent	Duration	Frequency	Reversibility	Environmental Consequence
Trivalent chromium in water column	negative	negligible	local	long-term	high	reversible (long-term)	low
Indirect effects of chromium via fish-food organisms	negative	negligible	local	long-term	high	reversible (long-term)	low

Note: Numerical score for ranking of environmental consequence is explained in Section 9.1, Table 9.5-1.

The environmental consequence to the north lake and northeast lake during post-closure is low

The residual impact classification (Table 9.5-35) for the post-closure exposure and resulting environmental consequences for fish health in the north lakes is low for direct effects of trivalent chromium in the water column and porewater, direct effects of copper in porewater, and direct and indirect effects of molybdenum in porewater. Although the impact magnitudes are negligible the environmental consequence is raised to low because of the long duration of the effect.

Low residual impacts are predicted for direct and indirect effects of chromium and aluminum in porewater and indirect effects of copper

The predicted environmental consequences from direct and indirect effects of chromium and aluminum in porewater and the indirect effects of copper in porewater were estimated to be low in both the north and northeast lakes. Effects would be sublethal and confined to less than 10% of the lake area in these lakes.

Elevated pH and nitrate in sediments are predicted to cause moderate environmental consequences

Elevated pH and nitrates in sediments have the potential to cause acutely lethal effects to fish. Therefore, moderate environmental consequences were predicted for up to 10% of the area for both the north and northeast lakes.

9.5.2.4.5 Level of Certainty in the Impact Assessment

There is a high level of certainty in the residual impact predictions for Snap Lake application and post-closure cases

Residual impact predictions for Snap Lake are based upon conservative, but well-calibrated water quality predictions and knowledge of baseline conditions. The conservative nature of the water quality predictions means that the risk of under-estimating the environmental consequences in Snap Lake is low. Furthermore, the knowledge of baseline conditions increases the certainty that the interpretation of predicted water quality effects on the aquatic biota was relevant and defensible.

Table 9.5-35 Residual Impact Classification for Impacts of Post-closure Groundwater Discharge to Fish Health in the North and Northeast Lakes

Chemicals of Concern	Direction	Magnitude	Geographic Extent	Duration	Frequency	Reversibility	Environmental Consequence
Trivalent chromium in water column (north lake only)	negative	negligible	local	long-term	high	reversible (long-term)	low
Trivalent chromium in porewater	negative	negligible	local	long-term	high	reversible (long-term)	low
Hexavalent chromium in porewater (north and northeast lakes)	negative	low	local	long-term	high	reversible (long-term)	low
Indirect effects of chromium via fish-food organisms	negative	low	local	long-term	high	reversible (long-term)	low
Copper in porewater	negative	negligible	local	long-term	high	reversible (long-term)	low
Indirect effects of copper via fish food organisms	negative	low	local	long-term	high	reversible (long-term)	low
Molybdenum in porewater	negative	negligible	local	long-term	high	reversible (long-term)	low
Indirect effects of molybdenum via fish food organisms	negative	negligible	local	long-term	high	reversible (long-term)	low
pH in porewater	negative	moderate	local	long-term	high	reversible (long-term)	low
Aluminum porewater	negative	low	local	long-term	high	reversible (long-term)	low
Indirect effects of aluminum via fish food organisms	negative	low	local	long-term	high	reversible (long-term)	low
Nitrate in porewater	negative	moderate	local	long-term	high	reversible (long-term)	moderate

Note: Numerical score for ranking of environmental consequence is explained in Section 9.1, Table 9.5-1.

There is a low level of certainty in the residual impact predictions for the north, northeast, NL5 and NL6 lakes post closure case; however, conservative assumptions are part of the assessment

Residual impact predictions for the post closure case conditions in the north, northeast, NL5, and NL6 lakes are uncertain. There are several sources of uncertainty. Water quality predictions are based upon the groundwater modelling. The groundwater model is uncertain because of the difficulty in predicting geochemical conditions, the lack of calibration data, and the uncertainty in the identification and characterization of groundwater flow paths. The water quality model is uncertain because of the lack of baseline data for use in calibration; however, this uncertainty is dealt with by using conservative assumptions of both groundwater flow volumes and concentrations of chemicals of concern. Water quality in NL5 and NL6 was not modelled. The effects assessment is uncertain because of the lack of baseline data on the aquatic communities in the four lakes. Again, to deal with the lack of baseline data, conservative assumptions regarding the types of organisms expected to be present and the potential for exposure were factored into the assessment of impact magnitudes.

The degree of conservatism in the post closure impact assessment for the north, northeast, NL5 and NL6 lakes is difficult to estimate

The overall degree of conservatism in the post closure impact assessment for the north, northeast, NL5, and NL6 lakes is difficult to determine. The difficulty is primarily related to the lack of baseline data on the fish community. For example, it is unknown if the area potentially affected by elevated chemical concentrations and pH in sediment porewater corresponds with critical fish habitat.

9.5.2.5 Key Questions F-4: What Impacts Will the Snap Lake Diamond Project Have on Fish Abundance?

9.5.2.5.1 Linkage Analysis for Snap Lake

Five linkages to fish abundance were analyzed

The following potential linkages between the Snap Lake Diamond Project and fish abundance in Snap Lake were analyzed:

- linkage between a change in access to fish resources for harvesting activity and a change to fish population abundance;
- linkage between blasting activity and fish population abundance;
- linkage between changes to fish habitat and fish population abundance;
- linkage between changes to fish health and changes to fish population abundance; and,
- linkage between individual changes to fish population abundance and changes to overall fish population abundance.

Fish Harvesting Activity

The linkage between fish harvesting and a change in fish abundance is invalid

Increased fishing pressure, as a result of increased access to a lake, can lead to substantial changes in the fish population structure of a lake (Munkittrick and Dixon 1989). Fish harvesting by mine personnel during all phases of the project will not be allowed. As such, the linkage between increased access leading to fish harvesting activity, and a subsequent change in fish population abundance, is not valid for the Snap Lake Diamond Project.

Blasting Activity

Detonation of explosives, or blasting, produces a shock wave that can harm free swimming fish and incubating fish eggs

Blasting, or explosives detonation, will be used as part of the underground mining process as well as during quarrying activities on the northeast peninsula. A consequence of blasting is a shock wave that radiates outward from the point of detonation. This may produce post-detonation compressive shock waves that would cause "a rapid rise to a high peak pressure followed by a rapid decay to below ambient hydrostatic pressure" (Wright and Hopky 1998). The drop below ambient hydrostatic pressure causes most of the negative effects on fish (Wright and Hopky 1998), which can range from damage to the swimbladder or other organs (*e.g.*, kidney, liver, or spleen), to the disruption of development and mortality of fish eggs. Small fish are more susceptible to the effects of shock waves than large fish (DFO 1982). Changes in fish behaviour have also been observed in relation to exposure to shock waves (Wright and Hopky 1998).

The pathway between blasting and a change to fish population abundance is a valid pathway

The areas associated with the proposed mine where free-swimming fish may be affected would be in open areas of Snap Lake, above the underground blasting sites, and in inland lakes with fish adjacent to any quarrying activity. Fish eggs could be affected if they were deposited on any shoals or shoreline areas located near enough to the blasting locations that they would be influenced by shock waves generated. Based on this, the pathway between blasting and a change to fish population abundance is a valid pathway.

Fish Habitat

The linkage between habitat and fish population abundance in Snap Lake is invalid

Based on the impact assessments in Key Questions F-2 and F-3, a negligible magnitude of impact was predicted for all potential pathways linked to fish habitat in Snap Lake. As such, the linkage between changes to fish habitat and fish population abundance is invalid.

Fish Health

The linkage between an effect on fish health and population abundance in Snap Lake is invalid

A direct effect on fish health has the potential to effect fish population abundance. Key Question F-3 evaluated the direct and indirect effect of the Snap Lake Diamond Project on fish health in Snap Lake. A negligible magnitude of impact was predicted for all potential pathways linked to fish health in Snap Lake. Based on this, the linkage between fish health and a change in fish populations in Snap Lake is not a valid pathway.

Overall Fish Abundance

Overall fish abundance is considered when two or more individual impacts may be present

Potential impacts to lakes within the LSA have been identified as single (*e.g.* Snap Lake and lakes NL5 and NL6) and multiple (north and northeast lakes). When two or more potential individual impacts to fish abundance have been predicted, their overall additive or synergistic effect on overall fish abundance will be considered separately from individual impacts under the heading of overall fish abundance.

There are no multiple impacts to fish in Snap Lake

In the case of the Snap Lake Diamond Project, the only linkage to consider, in terms of fish abundance, is the potential impacts of blasting. Consequently, this linkage is valid for an individual effect in Snap Lake and invalid for multiple effects.

9.5.2.5.2 Linkage Analysis for the North Lakes

The following potential linkages between the Snap Lake Diamond Project and fish abundance in the north lakes were analyzed:

- linkage between a change in access to fish resources for harvesting activity and a change to fish population abundance;
- linkage between blasting activity and fish population abundance;
- linkage between changes to fish habitat and fish population abundance;
- linkage between changes to fish health and changes to fish population abundance; and,
- linkage between individual changes to fish population abundance and changes to overall fish population abundance.

Fish Harvesting Activity

The linkage between fish harvesting and a change in fish population abundance is invalid

Mine staff will not have access to the north lakes at any time. In addition, fish harvesting by mine personnel during all phases of the project will not be allowed. As such, the linkage between increased access leading to fish harvesting activity, and a subsequent change in fish population abundance, is not valid for the proposed project.

Blasting Activity

Blasting is not linked to fish abundance

There will be no blasting in the vicinity of the north lakes. The pathway between blasting and a change to fish population abundance is not a valid pathway.

Fish Habitat

The linkage between fish habitat and fish population abundance, post-closure, in the north lake, northeast lake, NL5 and NL6 is valid

Based on the residual impact classification presented in Table 9.5-32 (Key Question F-2), three potential impacts to fish habitat were evaluated as having an environmental consequence greater than negligible. These included changes to the food source in the north lake, northeast lake, and a reduction in flow to the north lake outlet stream. The potential impacts of the changes in food source in the north lake and northeast lake to fish habitat were assessed in Key Question F-2. The linkage between the reduction in flow to the north lake outlet stream and an effect to fish population abundance is a valid pathway as a single impact to this stream.

Health of Fish in NL5 and NL6

The linkage between fish health and fish population abundance, post-closure, in NL5 and NL6 is invalid

The residual impact classification for potential effects to fish health, post-closure, in NL5 and NL6 considered two potential impacts related to trivalent chromium in the water column and indirect effects of chromium via fish-food organisms (Table 9.5-34). Both were predicted to have negligible environmental consequence. Consequently, the linkage between an effect to fish health and an effect to overall fish population abundance in the NL5 and NL6 is invalid.

Health of Fish in North and Northeast Lakes

The linkage between fish health and fish population abundance, post-closure, in the north and northeast lakes is valid

The residual impact classification for potential effects to fish health, post-closure, in the north and northeast lakes identified seven impacts with low or moderate environmental consequence (Table 9.5-35). These include the indirect effects of chromium, copper and aluminum via fish-food organisms and the direct effects of trivalent chromium in the water column and hexavalent chromium, pH, aluminum and nitrate in porewater. Based on this information, the linkage between an effect to fish health and an effect to fish abundance in the north lakes is a valid pathway.

Overall Fish Abundance

The linkage between individual impacts on fish abundance and overall fish abundance, post-closure, in the north and northeast lakes is valid

The combined effects of the residual impacts for fish health, at post-closure, in the north and northeast lakes identified in Key Questions F-3 on overall fish abundance must be considered. Based on the presence of multiple impacts on the north lakes, the linkage between an effect to fish health and an effect to overall fish abundance in the north lakes is a valid pathway.

9.5.2.5.3 Impact Analysis

Snap Lake

Blasting

Department of Fisheries and Oceans guidelines for blasting activity exist for the protection of fish

Wright and Hopky (1998) developed draft guidelines for the protection of fish from exposure to shock waves induced by the use of explosives. Two factors occur as a result of the use of explosives that may cause harm to fish and fish eggs: an instantaneous pressure change or "overpressure"; and "peak particle velocity", which is a measure of ground vibration. For adult fish, the threshold for an instantaneous pressure change is 100 kilo Pascals (kPa) (14.5 pounds per square inch [psi]) measured in the swimbladder of a free-swimming fish (all ages) (Wright and Hopky 1998). For spawning beds where eggs are incubating, peak particle velocities below 13 millimetres per second (mm/s) are recommended to ensure the protection and viability of the eggs (Wright and Hopky 1998).

Mining activity during production will be at least 170 m from the lake bottom

Based on the project description, underground blasting will occur generally at a frequency of once every 10 hours. The charge weight per delay during production is estimated to be approximately 227 kg per delay. The minimum distance below Snap Lake for blasting activity during advanced exploration was approximately 115 m and at the end of advanced exploration was approximately 170 m. During production this distance will continue to increase as mining activity moves further underground.

The shock wave produced by blasting depends on several factors

The intensity of a shock wave produced by explosives will vary depending on the weight of the charge used and the time delay between explosions. The attenuation of shock waves will also vary depending on several factors, specifically:

- the weight of each charge (kg);
- the distance between the point of the explosion and potential fish habitat; and,
- the characteristics of the substances through which the explosive shock waves are travelling.

During the 2001 blast monitoring program, both overpressure and peak particle velocity were below guideline limits

A blast monitoring program was conducted on Snap Lake in July 2001 (Appendix IX. 13). During this program, ground vibration and overpressure recording equipment was set-up at a number on stations on and around Snap Lake. The distances from the blasts range from 154 m away, at the recording station on Snap Lake directly above the underground blast site, to 615 m away, on a known lake trout spawning shoal south-east of the blast location. The maximum recorded overpressure during this monitoring program was 7.0 kPa at the site directly above the underground blast location. The maximum particle velocity (peak ground vibration) recorded on the known spawning shoal was 1.3 mm/s. These recorded blasting activity results are below the guideline limits for the protection of free-swimming fish and incubating fish eggs.

Predicted overpressure and peak particle velocities were calculated for several charge weights and several distances from blast locations

Based on the results of the monitoring program and the expected blasting activity, overpressure and ground vibration values were calculated for the proposed project at varying distances from underground blasting (Table 9.5-36). These calculations are based on a conservative rock density of 2.7 grams per cubic centimetre (g/cm^3) in granitic rock. The rock density under Snap Lake ranges between 2.7 g/cm^3 and 3 g/cm^3, based on laboratory testing of samples of the underground mine rock.

Both overpressure and peak particle velocity are within guideline limits 170 m from the blast location with the proposed production charge weight of 227 kg

At the estimated production charge weight of 227 kg (500 pounds [lbs.]) per delay, both overpressure and ground vibrations are below the guideline limits on the lake substrate immediately above the blast location. At a charge weight of 454 kg (twice the expected production charge weight), the guideline for ground vibrations is marginally exceeded immediately above the blast location; however, no known fish spawning habitat was identified in this area. At a distance of 250 m away from a 454 kg charge, both overpressure and ground vibrations are well below guideline thresholds. No known fish spawning habitats are present within 250 m of the proposed underground blasting areas.

Table 9.5-36 Calculated Overpressure and Ground Vibration Values for Varying Distances from Underground Blasting Activity

Set-back Distance	Physical Parameter	Charge Weight per Delay (kilograms)		
		86	227	454
100 m	overpressure	18.6	34.1	52.7
	ground vibrations	13.9	25.9	40.3
170 m	overpressure	6.9	12.6	19.5
	ground vibrations	5.1	9.4	14.7
250 m	overpressure	3.4	6.2	9.6
	ground vibrations	2.4	4.5	7.0
500 m	overpressure	0.9	1.7	2.6
	ground vibrations	0.7	1.2	1.9

1 Max Charge weight per delay from July 22, 2001 blast No. 9.
2 500 lbs. per delay for future production rounds estimated from AMEC.
3 Value for 1000 lbs. per delay.

Guideline limits will not be exceeded as the result of quarry blasting, based on the locations of the quarries and assumed charge weights

Quarrying activity for the proposed project will occur within the boundaries of the north pile and quarry developments will be located a distance greater than 1 km from any fish bearing waterbodies. The use of confined explosives (*i.e.*, charges set within any substrate as per Wright and Hopky 1998), with charge weights less than those proposed for the underground mining activity, at a distance greater than 1 km from any fish bearing waterbodies, will not result in the exceedance of any of the guideline limits.

North Lake Outlet Stream

Changes in volumes of flow from outlet stream from the north lake to NL5 and NL6 are predicted to occur during the operations period and could affect fish habitat

Based on the reduction of groundwater flow to the north lake, a reduction in outflow from this lake is predicted. This reduced outflow will also be carried forward as a reduced outflow in the stream leading to NL5 and NL6, and the northeast lake downstream (Section 9.3.2). For the outlet stream from the north lake the maximum reduction in discharge is predicted to be approximately 8% at the end of the operations period. During most of the operations period the reductions in discharge will be less than this (*i.e.*, a gradual increase in effect will occur as the underground operations expand to a maximum level). These reductions in outlet stream discharges will cease once groundwater flow returns to baseline conditions at the end of the operations period. In the Hydrology Impact Assessment (Section 9.3.2) the environmental consequence of these flow reductions was rated as low for the north lake.

The impact to fish habitat in the outlet streams from the north lake in the local study area will be a maximum of an 8% reduction in flows

Currently, little is known about the nature of the outlet stream from the north lake to lakes NL5 and NL6. Because the reduction of flows will occur throughout the year, but in varying magnitude, there are several potential impacts. These include reduced spawning habitat, reduced rearing habitat, and reduced migratory capacity that may result in stranding of juvenile and adult fish in areas incapable of supporting overwintering (*e.g.*, in NL5 or NL6, or areas of the stream channel). Based on the discussion presented in Section 9.3.2, it is likely that the greatest proportional change in outflow will occur during low flow periods (*i.e.*, during winter and late summer or fall base flows) with relatively smaller reductions in flow during the high spring run-off period. A reduction in flows by 8% would still be considered to be within the historical range of low flow events.

The impact to fish habitat in the outlet stream from the north lake in the local study area has been assessed as having a low magnitude

The north lake is a headwater lake with a limited catchment area. As such, flows through the outlet stream are expected to increase in spring and early summer and decrease substantially as the summer progresses towards fall and winter. The large fish species most likely to utilize the habitats in either the outlet stream of the north lake or lakes NL5 and NL6 are Arctic grayling and longnose sucker. Both these species are known to use Arctic streams in spring for spawning and rearing habitat and in summer for foraging habitat.

This period corresponds with the lowest proportional change in outflow predicted for the outlet stream. Natural low flows in fall and winter would deter Arctic grayling and longnose sucker from utilizing the type of habitat found in the outlet stream and lakes NL5 and NL6 for overwintering or foraging. Since this corresponds to the greatest proportional change in outflow, which at its maximum will only reach 8%, the impact magnitude is rated as low. The ecological consequence of this change in flow is rated as low since the lowest proportional change in outflow corresponds with the highest potential for fish to utilize the habitat. In addition, the highest reduction in outflow from the north lake will only occur for a relatively short period near the end of operations. For the majority of the duration of this outflow reduction, flows will be decreased by less than 8%.

North and Northeast Lakes

The importance of the individual residual impacts to overall fish abundance will be assessed for the north and northeast lakes

The individual residual impacts greater than negligible at post-closure for fish health in the north lake and northeast lakes will be assessed in terms of their importance to overall fish abundance. A summary of the residual impacts identified for the north lake and northwest lake is listed in Table 9.5-37. These are based on the impact analysis completed in Key Questions F-1 and F-3.

Table 9.5-37 Summary of Residual Impacts to Fish in the North and Northeast Lakes

Source of Impact	Chemicals of Concern	Magnitude	Environmental Consequence	Spatial Extent	Receptor
Direct effects	pH in porewater	moderate	moderate	1 – 10% of each lake	fish early life stages
Direct effects	nitrate in porewater	moderate	moderate	1 – 10% of each lake	fish early life stages
Direct effects	aluminum porewater	low	low	1 – 10% of each lake	fish early life stages
Direct effects	hexavalent chromium in porewater	low	low	1 – 10% of each lake	fish early life stages
Indirect effects	chromium via fish-food organisms	low	low	1 – 10% of each lake	phytoplankton, zooplankton, and benthos
Indirect effects	copper via fish food organisms	low	low	1 – 10% of each lake	phytoplankton, zooplankton, and benthos
Indirect effects	aluminum via fish food organisms	low	low	1 – 10% of each lake	phytoplankton, zooplankton, and benthos

Summary of Impacts from Key Question F3.

All impacts for the north and northeast lake occur within a restricted area

For all of the residual impacts listed in Table 9.5-35 it must be remembered that they all occur within the same restricted area of either the north or northeast lake. Therefore, the impact to overall fish abundance is occurring in <10% of the total area of the lakes.

The northeast lake will receive 50% less groundwater discharge than the north lake

During the course of the impact assessment in Key Questions F-1, F-2 and F-3, a conservative estimate for the spatial extent of the impact of between 1 and 10% was used. However, the groundwater model shows that the northeast lake will receive 50% less mine influenced groundwater (200 cubic metres [m^3]) than the north lake (400 m^3) at post-closure (Section 9.2, Hydrogeology). The area affected by groundwater discharge in the northeast lake will be a very small portion of the western corner of the lake.

Porewater pH is expected to reach 11.8 and has the potential for acute, localized, effects on fish early life stages and benthic organisms

Key Question F-3 identified that the predicted porewater pH of 11.8 is expected to be lethal to some benthic aquatic organisms and early life stage fish. In addition, elevated concentrations of nitrate in the affected porewater have the potential to be converted to ammonia at a pH of 11.8. This elevated pH and ammonia zone effect is limited to <10% of the lake. The impact magnitude of pH effects on fish health was rated as moderate because of the localized potential for direct mortality and indirect effect to food organisms.

A measurable change in fish populations in the north lake, as a result of pH, is predicted, however only a limited area of the lake is effected

Based on the same assumptions and evaluation presented for chromium effects, we would expect a measurable change in fish population abundance and condition. However, we would not expect the level of change resulting from increased pH, and associated ammonia, to compromise the long-term sustainability of the fish populations in the affected lakes. This is based on the continued availability of spawning and rearing habitat in other areas of the lake and the continued availability of food in the lake. As a result, the impact magnitude at the fish population level is rated as low for direct and indirect pH effects.

A measurable change in fish populations in the north lake, as a result of nitrate, is expected, however only a limited area of the lake is effected

The predicted nitrate concentration in the north and northeast lakes porewater is 42.7 mg/L. In addition to increasing ammonia, in a toxic form, in the porewater, this concentration exceeds some of the reported effects thresholds for early life stage fish. Therefore, there is a potential for effects on eggs and larvae via exposure to porewater concentrations of nitrate. Again this effect was predicted for <10% of the lake. Following the same rationale for potential population effects as above, the magnitude of the impact related to direct and indirect increased nitrate concentration is rated as low.

Increased chromium and aluminum concentrations have the potential for both direct and indirect effects on fish health and population abundance

Based on the discussion of chromium and aluminum presented in Key Question F-3, porewater affected by chromium will be above the HC_{20} benchmark in <10% of the lake. At the predicted concentrations, this represents a toxicity risk to early life stage fish. In addition, both chromium and aluminum are predicted to have an indirect, low impact on fish health through food organisms. However, impacts on fish food species are expected in less than 10% of the north and northeast lakes. There is considerable uncertainty in this prediction because it is unknown whether critical habitat areas will be affected by changes in porewater quality. Overall, the evaluation of impact potential has identified both direct and indirect effects to fish health from post-closure chromium and aluminum concentrations in the north lakes. Direct mortality is also a possibility for early life stages of fish directly in contact with the affected porewater areas. These potential impacts are, however, limited to <10% of the lake. It is also important to remember that there is a high level of uncertainty surrounding the groundwater chemicals of concern entering the north and northeast lake.

A measurable change in fish populations in the north lake, as a result of chromium and aluminum, is expected, however only a limited area of the lake is effected

Based on the above assumptions, we would expect a measurable change in fish population abundance and condition. However, we would not expect this level of change to compromise the long-term sustainability of the fish populations in the affected lakes. This is based on the continued availability of spawning and rearing habitat in other areas of the lake and the continued availability of food in the lake. As a result, the impact magnitude at the fish population level is rated as low for direct and indirect chromium and aluminum exposure.

The environmental consequence to fish populations is rated as low

Based on the information available, and several assumptions regarding the fish populations, the environmental consequence to fish populations from a change in fish health is predicted to be low. The potential for a post-closure change to water quality, and subsequently fish health and fish populations, was identified through the EA. There is a high level of uncertainty regarding the expected concentrations of concern. However, based on the information available, a measurable impact to fish populations is predicted and has been rated as having a low environmental consequence. The effect is within the LSA and has a long-term duration, but is reversible in the long-term as well. The low rating is based primarily on the limited spatial extent of the changes within the affected lakes. It is also based on assumptions regarding the types of fish habitat affected and the potential exposure of various life stages to direct mortality or reduced health, as well as indirect reduction in food quality.

The evaluation of population level impacts is related to changes in numbers, condition, and overall population sustainability

The key elements for evaluating the potential impact at the fish population level are an assessment of potential change to numbers, condition, and population sustainability. In addition to loss of critical habitat affecting fish populations, a large enough reduction in population numbers through mortality could result in the collapse of the population. Also, reduced health may have long-term consequences to reproductive potential and therefore, long-term survival at the population level.

As a result of the lack of baseline data, assumptions regarding the exposure were required

As there is no baseline information available for fish populations or habitats in the north lakes, several assumptions are required in order to assign an impact magnitude. We have no knowledge concerning the potential for fish spawning or rearing habitat to overlap with the substrate area affected by reduced porewater quality.

10% of the spawning and rearing habitat, and 10% of food supply will be lost

Therefore, for the north lake, we have to assume that on a proportional basis, the affected 10% of the lake represents 10% of the spawning and rearing habitat, and that this will be lost for the duration of the impact. We also have to assume uniform distribution of fish in the lake so that 10% of the adult population will be effected by reduced food quality and therefore reduced health. For the northeast lake, we can assume than considerably less than 10% of the lake will be affected to the same degree as the north lake since it will receive half the groundwater discharge. This evaluation also assumes that the predicted concentrations for chemicals of concern entering the north lakes are a conservative estimate given the information available during the preparation of the EA for the Snap Lake Diamond Project.

9.5.2.5.4 Residual Impact Classification

Snap Lake

Impact classification is provided in Table 9.5-38

Based on the impact analysis, Table 9.5-38 summarizes the residual impact and presents the environmental consequence for the potential effects to fish abundance in Snap Lake.

The environmental consequences of blasting on fish populations are predicted to be negligible

For both instantaneous pressure change and peak particle velocity, the magnitude was defined as negligible if the value for either factor has less than the guideline value, and high if the value exceeded the guideline. The DFO guidelines are expected to be met in all parts of Snap Lake and in all fish bearing inland lakes; therefore, the environmental consequences of blasting are negligible for the proposed project.

Table 9.5-38 Classification of Residual Impacts to Fish Population Abundance in the Snap Lake

Waterbody	Activity	Direction	Magnitude	Geographic Extent	Duration	Reversibility	Frequency	Environmental Consequences
Snap Lake	blasting	negative	negligible	local	medium-term	reversible (short-term)	medium	negligible
Outlet stream from north lake, lakes NL5 and NL6	groundwater flow reduction	negative	low	local	medium-term	reversible (short-term)	medium	low
North lake	groundwater effects to lake	negative	moderate	local	long-term	reversible (long-term)	continuous	moderate
Northeast lake	groundwater effects to lake	negative	low	local	long-term	reversible (long-term)	continuous	low

North Lake Outlet Stream

The consequence of outlet stream impacts is low

Based on the impact analysis, Table 9.5-38 summarizes the residual impact and presents the environmental consequence for the potential effects to fish abundance in the north lake outlet stream.

North and Northeast Lakes

Consequences to the north and northeast lakes are given

Based on the impact analysis, Table 9.5-38 summarizes the residual impacts and presents the environmental consequences for the potential effects to fish populations in the north and northeast lakes.

The environmental consequence to overall fish abundance from a change in fish health is predicted to be low for the northeast lake and moderate for the north lake

The potential for a post-closure change to water quality, and subsequently fish health and overall fish abundance, was identified through the EA. There is a high level of uncertainty regarding the expected concentrations of concern. There is also a high level of certainty that the environmental consequences assigned to each lake are the worst case scenario and will likely be less than predicted given the conservative nature of the predictions. Based on the information available, a measurable impact to fish populations is predicted and has been rated as having a moderate environmental consequence for the north lake. The low rating for the northeast lake is based primarily on the limited spatial extent of the changes within this lake. The effect is within the LSA and has a long-term duration, but is reversible in the long-term as well. Both ratings are based on assumptions regarding the types of fish habitat affected and the potential exposure of various life stages to direct mortality or reduced health, as well as indirect reduction in food quality.

Refinement to post-closure groundwater quality and flow predictions will be provided as supplemental information

The predicted rate of groundwater outflow from Snap Lake and the chemistry of the mine-affected groundwater used in the environmental assessment were based on conservative worst-case assumptions. Further refinement of groundwater flows and the chemistry of mine affected groundwater during post-closure is expected to reduce the magnitude of predicted impacts to the north and northeast lakes. Additional work is proposed or currently underway to refine the impact predictions for mine affected groundwater. The results of these investigations will be filed as supplemental information.

9.5.2.5.5 Monitoring

Blasting in Snap Lake

Additional blasting monitoring or a reduction of charge weight would be recommended, if blasting were to occur within 150 m of the lake bottom

There is no proposed long-term monitoring plan for blasting activity at the proposed project site. A follow-up monitoring program, once production blasting begins, would allow a refinement of the equations used to calculate peak particle velocity and overpressure. This would become more important if blasting were to occur within 150 m of any spawning habitat. At this distance, there is the possibility that peak particle velocity could be exceeded, depending on the charge weight. Consideration of charge weight should be a priority if blasting is within 150 m of the lake bottom. If either of the guidelines for instantaneous pressure change or peak particle velocity may be exceeded, an application may be submitted for authorization under the *Fisheries Act* for the use of explosives. Following a review of information concerning the need for an explosive event exceeding guidelines, an authorization may be issued by DFO for the use of explosives if it can be reasonably expected that there are no economically important and/or otherwise important biological resources at risk.

North Lakes

Baseline information is needed to prepare a monitoring plan

Detailed baseline information will be required for the north lake, northeast lake, NL5, NL6 and the outlet stream of the north lake in order to better evaluate the potential impacts to fish populations in the north and northeast lakes. Once this information is available, an appropriate monitoring plan can be prepared for these lakes. De Beers will refine the groundwater model to determine the true extent of these predicted changes. The level of baseline information required on the north lakes will depend on the results of the revised modelling.