DE BEERS

GROUP OF COMPANIES

May 1, 2014

File: L020

Rosanna Nicol
Mackenzie Valley Land and Water Board
PO Box 2130
Yellowknife, Northwest Territories
X1A 2P6

Dear: Ms. Nicol:

Re: De Beers Canada Snap Lake Mine
Aquatic Effects Monitoring Program (AEMP)
Items Water License #MV2011L2-0004

De Beers Canada Inc. (De Beers) would like to submit, the 2013 Aquatics Effects Monitoring Program to the Mackenzie Valley Land and Water Board under Part G of the Water License MV2011L2-0004.

A condition of the 2013 AEMP Design Plan approval was that De Beers submit a calculation of fish tissue normal range within the 2013 AEMP Annual Report. The normal range calculation is provided in Appendix 9A, and this condition has now been met.

Should you have any questions, comments or require further clarification, please do not hesitate to contact the undersigned at (867) 767-8646 or by e-mail at the following address: Alexandra.Hood@debeerscanada.com.

Sincerely,

DE BEERS CANADA INC.

A. Hood

Alexandra Hood
Permitting and Environmental Superintendent
Snap Lake Mine

Attachments

DE BEERS CANADA INC.

SNAP LAKE MINE

AQUATIC EFFECTS MONITORING PROGRAM

2013 ANNUAL REPORT

May 2014

PLAIN LANGUAGE SUMMARY

Introduction

De Beers Canada Inc. (De Beers) owns and operates the Snap Lake Mine (the Mine), a diamond mine located approximately 220 kilometres (km) northeast of Yellowknife, Northwest Territories (NWT). The Aquatic Effects Monitoring Program (AEMP) is designed to monitor Snap Lake for Mine-related effects, to verify and update the Environmental Assessment Report (EAR) predictions, and to inform management decisions made by the Mine. The AEMP fulfills the requirements of Part G of Water Licence MV2011L2-0004 for the Mine. Components of the AEMP must also comply with Part F of the Water Licence. The Mackenzie Valley Land and Water Board (MVLWB) approved the AEMP in July 2005 and an updated AEMP Design Plan in November 2013. The final 2013 AEMP Design Plan was submitted to the MVLWB in January 2014. This document represents the 10th annual AEMP report for the Mine and presents the results of the 2013 program. This is the first annual report under the 2013 AEMP Design Plan.

The core of the AEMP, monitoring of water quality, plankton, sediment quality, benthic invertebrates, fish health, fish tissue chemistry, and fish community are undertaken on an annual basis. Other components, such as the fish community monitoring, tissue chemistry, and fish health are undertaken every three years. Fish tissue chemistry and fish community monitoring were conducted in 2013. Special studies conducted in 2013 include the Littoral Zone Special Study, Picoplankton Special Study, Downstream Lakes Special Study, Lake Trout Population Estimate Special Study, and Stable Isotope Food Web Analysis Special Study.

Site Characterization and Environmental Variables

The Site Characterization and Supporting Environmental Variables chapter summarizes information to describe the general conditions at the Snap Lake Mine site and the local environment in which the Aquatic Effects Monitoring Program (AEMP) is conducted. The data presented in this chapter will assist in the interpretation of the component-specific AEMP results by the main AEMP components (i.e., water quality, sediment quality, plankton, benthic invertebrates, and fish community).

Three reportable spills occurred at the Snap Lake Mine site; however, these spills did not enter Snap Lake, and are not anticipated to have a measurable effect on the aquatic environment. Effluent discharge rates have increased with a 28% higher effluent discharge in 2013 than in 2012. Rainfall was approximately 3.6% higher than in 2012, but followed the same seasonal pattern as in previous years. Average annual air temperature was approximately 1.5°C colder than in 2012. Wind strength and pattern were similar to previous years. The water elevation of Snap Lake increased by approximately 0.062 m between 2012 and 2013 and remained within the range of elevations surveyed between 2002 and 2012. Snap Lake had a lower range of elevation changes between 2002 and 2013 than the 1999 Reference Lake.

Snap Lake and both reference lakes, Northeast Lake and Lake 13 showed a similar pattern in temperatures with variability due to wind-induced mixing; temperatures in the upper layers increased from mid-July to mid-August, and then decreased into September. Thermoclines were observed at deep locations in late summer. Ice thickness and cover was within the range of measurements from previous sampling years.

Water Quality

The water quality component (Section 3) summarizes all data obtained from water samples and field measurements collected from Snap Lake in 2013. Over 200 water samples were collected from Snap Lake and surrounding waterbodies (i.e., reference lakes [Northeast Lake and Lake 13], inland lakes, Streams S1 and S27, and upstream of King Lake). In addition, water samples were collected for the Downstream Lakes Special Study (Section 11.3) and the Nutrient Assessment (Appendix 3B).

Samples were shipped to analytical laboratories across Canada to obtain chemical analyses. The water quality results were compared to Water Licence limits, Aquatic Effects Monitoring Program (AEMP) benchmarks, site-specific water quality objectives, drinking water guidelines, environmental assessment predictions, updated predictions, and data from previous years. Water quality results from Snap Lake and the Snap Lake Mine's water intake were also compared to Canadian drinking water quality guidelines to assess the drinkability of Snap Lake water.

The daily volume of effluent discharged to Snap Lake from the Mine has increased since 2004, when treated effluent discharge began, with consequent increased loadings to the lake. In 2013, the annual treated effluent volume was approximately 31% higher than in the 2012 AEMP reporting year.

Some water quality parameters have increased in Snap Lake since the Mine started operating. Concentrations of total dissolved solids (dissolved salts in the water), nutrients (specifically nitrogen), and some metals have increased in Snap Lake related to treated effluent discharged from the Mine. Concentrations of nitrate, chloride, and fluoride were above an AEMP benchmark on at least one occasion in 2013. However, increases in these parameters were accompanied by increased hardness, which is a parameter that reduces the toxicity of those parameters. All 2013 concentrations of nitrate, chloride, and fluoride were below recommended site-specific objectives and predicted values available from the EAR. Treated effluent and receiving waters were not toxic based on laboratory toxicity testing.

Concentrations of most water quality parameters in Snap Lake were below health-based drinking water guidelines, with the exception of *Escherichia coli* (*E. coli*) and total coliforms. Microbiological parameters can naturally exist in the aquatic environment. Drinking water at the Mine is filtered and chlorinated prior to consumption (as required by Health Canada of any surface waters in Canada); treated drinking water quality was acceptable from a microbiological perspective (*E. coli* and coliforms). Drinking water at the Mine will continue to be tested regularly and the results reported to the local Health Authority.

The Mine's initial environmental assessment predicted that concentrations of water quality parameters associated with the treated effluent discharge would reach near background concentrations within 44 kilometres (km) downstream of Snap Lake. In 2013, concentrations of Mine-related parameters reached background concentrations approximately 11 km downstream of Snap Lake.

Treated effluent discharge from the Mine is increasing and, as a result, water quality is changing in Snap Lake as predicted. However, based on the 2013 data, including toxicity testing, the changes to water quality in Snap Lake are unlikely to result in adverse effects to resident aquatic life, nor to affect the drinkability of Snap Lake water.

Sediment Quality

Sediment quality was monitored at the Snap Lake diffuser station and at five stations in one reference lake, Lake 13, in 2013 (Section 4). This was the first year of monitoring under the 2013 Aquatic Effects Monitoring Program (AEMP) Design Plan, which calls for monitoring the Snap Lake diffuser station annually and the full set of AEMP stations every three years. Lake 13 was sampled to assess unusual results reported in 2012. Sediments were analyzed for particle size distribution, total organic carbon (TOC) content, nutrients, and metals. Average concentrations of a number of metals were higher in Lake 13 than in Snap Lake, indicating that concentrations of some metals are naturally elevated in this geographic region. The 2013 results for Lake 13 confirmed the presence of higher concentrations of arsenic, barium, cobalt, iron, and manganese at one station in the northeast area of the lake, similar to the 2012 results.

Sediment quality in Snap Lake was only assessed at the diffuser station in 2013; trends over space and time in the main basin of the lake were not evaluated. Sediment quality was compared in the top 5-centimetre (cm) and top 2-cm layers of sediment to see whether there were differences in more recently deposited sediments; comparisons were conducted over time and to baseline conditions. The results of these comparisons indicated that concentrations of available sulphate, calcium, mercury, sodium, and strontium at the diffuser stations are potentially being influenced by Snap Lake Mine (Mine) operations. However, these changes are unlikely to have resulted in adverse environmental effects.

Plankton

The plankton component of the Aquatic Effects Monitoring Program (AEMP) report (Section 5.0) evaluated whether there were any changes happening to the small plants (phytoplankton) and animals (zooplankton) in Snap Lake water due to nutrients added by the Snap Lake Mine (Mine). These small plants and animals are together referred to as plankton. Changes in plankton can affect fish in the lake because plankton are part of the food chain upon which fish rely. Such changes can happen before fish are affected. In 2013, plankton were collected at nine locations in Snap Lake (five in the main basin of the lake where the Mine is located and four in the northwest arm), once in each of July, August, and

September. Plankton were also evaluated at ten locations in two reference lakes not affected by the Mine. Five locations were evaluated in Northeast Lake and five locations in Lake 13.

Nutrient concentrations have changed in Snap Lake since 2004 when the Mine started operating. Nitrogen and silica concentrations are increasing in the lake but phosphorus concentrations have not changed. Until 2011, there were more small plants in the main basin of Snap Lake compared to the northwest arm of Snap Lake; in 2011 and through to 2013, the number of small plants increased in the northwest arm but decreased in the main basin. There were similar amounts of small plants in Snap Lake, Northeast Lake, and Lake 13 in 2013, so nutrients did not have a large effect on the amount of small plants. However, the types of small plants in Snap Lake may be affected by the nutrients in the lake, because the different types of small plants have changed since 2004. The small animals in Snap Lake have decreased in numbers from 2004 to 2013, and the different types of small animals within Snap Lake have changed. Small changes are happening in Snap Lake. These changes may become greater with continued input of nutrients from the Mine. At present, both the small plant and animal communities in Snap Lake are healthy.

Benthic Invetebrate Community

The benthic invertebrate section of the AEMP report (Section 6) evaluated whether the discharge of treated effluent has caused changes in the numbers and types of small animals that live on the bottom of Snap Lake. These animals are referred to as benthic (bottom-dwelling) invertebrates (animals without backbones), and include snails, clams, worms, and insects. These organisms provide food for fish. Changes in the numbers and types of bottom-dwelling invertebrates can cause changes in the numbers and types of fish in the lake.

Lake bottom sediments were collected in fall 2013 from ten locations in Snap Lake, five locations in Northeast Lake, and five locations in Lake 13. The invertebrates in sediment collected at these locations were identified and counted. The numbers and types of invertebrates were then compared between Snap Lake and the two reference lakes combined (Northeast Lake and Lake 13), between the two reference lakes, and between Snap Lake and Northeast Lake. The numbers of invertebrates varied widely in all lakes. There were differences between Snap Lake and the two reference lakes combined, and also between the two reference lakes. This tells us that most of the differences among the lakes resulted from differences between Northeast Lake and Lake 13, in other words between the two reference lakes. The differences in the benthic community in Lake 13 compared to both Northeast Lake and the main basin of Snap Lake show that Lake 13 is not a suitable lake for direct comparisons to the main basin of Snap Lake. There were few differences between the established reference lake (Northeast Lake) and Snap Lake that could have been caused by the Mine discharge. Numbers of the non-biting midge (*Micropsectra*) were lower in the main basin of Snap Lake compared to Northeast Lake. Also, numbers of a snail (*Valvata sincera*) and another non-biting midge (*Tanytarsus*) were higher in the main basin of Snap Lake compared to Northeast Lake. The benthic community also changes over time naturally and the changes observed to date in Snap Lake are generally within the normal range for Northeast Lake.

The benthic invertebrate community in Snap Lake remains healthy. Community variables remain within ranges that are considered normal, based on data from baseline studies and monitoring in Northeast Lake, except for the fingernail clams, which were slightly more abundant than this range. The overall effect of Mine discharge on the benthic invertebrate community has to date been low and within the range predicted in the initial environmental assessment for the Mine.

Fish Health

This section (Section 7) was not required as part of the 2013 AEMP; the Snap Lake fish health was last sampled in 2012 and will next be sampled in 2015. This section is maintained as a placeholder for reporting in the 2015 Annual Report as detailed in Section 1.5.

Fish Community Monitoring

In 2013, De Beers conducted the second standard fish population monitoring program to collect data necessary to monitor potential changes in fish populations associated with the Snap Lake Mine (Mine). The program will be implemented every three years for the life of the mine. Fish population monitoring was completed in Snap Lake and two reference lakes, Northeast Lake and Lake 13, in July 2013 to answer the key question: will the fish community in Snap Lake be affected by Mine-related changes in water quality in Snap Lake and will any change be greater than predicted in the Environmental Assessment Report (EAR). No change to the fish community was predicted in the EAR.

A Broad-scale Monitoring (BsM) gill net program was used to collect fish in the three lakes. A total of 460 individuals were captured. The numbers of Lake Trout and Round Whitefish appeared higher in Snap Lake than the reference lakes. Slight variations were observed in the size and age of Lake Trout and Round Whitefish among lakes. The composition of the fish community in Snap Lake did not appear to change between 1999 and now including Arctic Grayling, Burbot, Lake Chub, Lake Trout, Longnose Sucker, and Round Whitefish. One species, Slimy Sculpin, was not captured in 2013 in any study lake that was captured before; the gill net method did not capture them. In future, other methods will be added to try to catch Slimy Sculpin.

In conclusion, fish appeared healthy and abundant in Snap Lake. There were some differences in fish between Snap Lake and the reference lakes, and these differences could be natural resulting from differences in lake temperature, or due to the netting method. Based on results of this study, there have been no changes to the fish community composition of Snap Lake that could be attributed to Mine-related changes in water quality. This program provided data about fish that will be useful in monitoring future changes over time.

Fish Tissue Chemistry

Fish tissue chemistry (Section 9) was completed on large-bodied fish in 2013. Lake Trout and Round Whitefish were sampled in 2013 from Snap Lake, Lake 13, and Northeast Lake to determine their tissue chemistry. The sampling program was expanded from earlier programs to include kidney and liver as well as muscle tissue.

Results showed that two metals were increasing in muscle tissue in Snap Lake: thallium and cesium. These metals were elevated relative to the baseline in Snap Lake, the reference lakes, and were also above the range of natural variability in the region, known as the 'normal range'. These increases in metal concentrations were observed in all tissue types, including liver, kidney, and muscle tissues. However, it was uncertain how these increased metal concentrations were connected to the Snap Lake Mine (Mine).

Five additional metals were detected in higher concentrations in fish tissues from Snap Lake in 2013 compared to the reference lakes and were above the range of natural variability in the region in either liver, kidney or muscle tissue: iron, mercury, molybdenum, potassium, and strontium. Potassium and strontium concentrations were also elevated in muscle tissue relative to the baseline in Snap Lake. Fourteen metals were detected at lower concentrations in fish tissues from Snap Lake in 2013 compared to the reference lakes and were below the range of natural variability in the region: aluminum, arsenic, barium, bismuth, cadmium, calcium, copper, magnesium, manganese, nickel, phosphorous, rubidium, silver, and zinc. Arsenic, calcium, and rubidium concentrations were also below baseline concentrations in muscle tissue. Because of inconsistent changes in metal concentrations across tissue types and species, and the absence of differences from baseline in Snap Lake for all metals except arsenic, calcium, and rubidium, these changes in metal concentrations were determined to not be connected to the Mine.

There were no fish tissue samples above Canadian Food Inspection Agency commercial consumption guidelines for arsenic or lead in Lake Trout or Round Whitefish tissues in 2013. Some Lake Trout from each of Snap Lake, Northeast Lake, and Lake 13 had kidney, liver, and muscle mercury concentrations above the commercial consumption guideline for mercury, which was also seen in fish prior to the start of Mine operations. Only one Round Whitefish had a liver tissue mercury concentration above the commercial consumption guideline; such exceedances occur naturally and were determined to not be connected to the Mine.

There were no fish health or fish taste issues raised during the 2013 annual fish tasting. There is no elevated risk to traditional fishers in consuming fish from Snap Lake relative to other lakes in the region.

Fish Tasting

Fish tasting is conducted annually by De Beers. Fish tasting is an informal, annual gathering of members of Aboriginal organizations and De Beers staff at the Mine site to taste fish from Snap Lake. In 2013, 17 fish were captured and one was released. Sixteen fish were prepared, and evaluated. Overall, Aboriginal community members agreed that the health, and taste of the fish from Snap Lake ranged from good to excellent. Community members commented that there were 'good fish in these lakes.'

Littoral Zone Special Study

A Littoral Zone Special Study (Section 11.1) was initiated in 2012 to determine the feasibility of sampling the near-shore areas of Snap Lake and Northeast Lake. Littoral zone sampling was completed in August 2012 and 2013, and will continue in 2014.

The littoral zone is the shallow near-shore area of lakes. Snap Lake and Northeast Lake have large littoral zones, accounting for close to half of the total areas of these lakes. Unlike the deeper open-water area of a lake, the littoral zone provides habitat for small plants (algae), animals without backbones (invertebrates; e.g., snails, worms, insects), and fish to live. When nutrients are added to the lake water, algae can grow faster and provide more food for invertebrates and fish in the littoral zone.

Food quality for littoral invertebrates was poorer in Northeast Lake compared to Snap Lake, and nutrient concentrations in the littoral zone of Snap Lake were higher in 2012 and 2013 compared to 2004, when a preliminary assessment was completed before mining started. This may mean more food is available for invertebrates and fish in Snap Lake because of the nutrients discharged from the Snap Lake Mine. The amount of algae was higher in the littoral zone of the main basin of Snap Lake in 2012 and 2013 compared to 2004 and higher in the littoral zone of Snap Lake compared to the littoral zone of Northeast Lake. But the amount of material (bacteria, and other living and dead organisms) on the rocks in the littoral zone of Northeast Lake was higher than in the littoral zone in Snap Lake. The types of algae also differed between sampling years (2004, 2012 and 2013) in Snap Lake, and between Snap Lake and Northeast Lake. Similarly, the types of littoral invertebrates collected in 2013 were different between Snap Lake and Northeast Lake.

The Littoral Zone Special Study demonstrates that littoral zone monitoring is possible in Snap Lake and Northeast Lake, and the improved sampling methods in 2013 have allowed for the collection of more reliable data compared to 2012.

Picoplankton Special Study

The Picoplankton Special Study, initiated in 2008, supports the plankton component (Section 5) of the Aquatic Effects Monitoring Program (AEMP). It evaluates whether there were any changes happening in certain bacteria and small plants that are part of the “microbial loop”, which is a model of pathways for nutrient and carbon cycling by microbial components in the open-water community. This study helps examine changes in the large small plants called phytoplankton. Together, these results can help answer the question of whether changes in Snap Lake waters are due to nutrients or other substances added by the Mine. Changes in picoplankton and other plankton can affect fish in the lake because they are part of the food chain upon which fish rely. Such changes can happen before fish are affected.

The data suggest Mine-related nutrient enrichment within Snap Lake, although other factors (e.g., increasing total dissolved solids) may also be affecting the picoplankton. The changes observed are subtle and may not affect the food chain upon which fish rely. Quantitative comparisons (i.e., statistical tests) will be completed as part of the AEMP four-year re-evaluation report in 2016.

Downstream Lakes Special Study

Daily discharge rates from the Snap Lake Mine (Mine) have steadily increased since discharge began in 2004, resulting in changes to water quality in Snap Lake. Treated effluent is becoming evenly mixed throughout the main basin of Snap Lake and, as predicted, is present in lakes downstream of Snap Lake. Results from monitoring programs conducted in 2011 and 2012 showed evidence of treated effluent, elevated dissolved salts and nutrients, throughout the first two small lakes immediately downstream of Snap Lake and within 50 metres (m) and 650 m of the inlet of Lac Capot Blanc, respectively.

Based on results from the 2011 and 2012 monitoring programs, it was recommended that further information be collected in the first three downstream lakes (i.e., DSL1, DSL2, and Lac Capot Blanc). Accordingly, the Downstream Lakes Special Study was completed in May, July, August, and September of 2013 to collect additional information (i.e., bathymetry, supporting environmental variables, water and sediment quality, and chlorophyll) from these lakes and to further document the extent of treated effluent downstream of Snap Lake.

In 2013, treated effluent was evident in DSL1, DSL2, and Lac Capot Blanc. Concentrations of dissolved salts and nutrients decreased with distance downstream. The extent of the effluent plume was observed approximately five kilometres (km) from the inlet of Lac Capot Blanc, which is farther from the inlet than in 2012. The treated effluent mixed rapidly as it entered Lac Capot Blanc (as evidenced by a notable decrease in salt and nutrient concentrations near the inlet) then dispersed gradually, with concentrations returning to background levels within approximately five km of the inlet. Concentrations of most water quality parameters were below guidelines for the protection of aquatic life, with the exceptions of pH in the ice-covered season, and lead and fluoride at two stations in the open-water season. Sediment from the three downstream lakes was comprised mainly of fines, and the sediment quality was comparable to Snap Lake and Northeast Lake.

The 2014 downstream sampling program will continue to collect information on the extent of the treated effluent plume, and on water and sediment quality. Biotic sampling, including benthic invertebrate, plankton, and fisheries programs, will be completed in 2014.

Lake Trout Population Estimate Special Study

A mark recapture study was conducted to: answer the Key Question "How many Lake Trout of fishable size (greater than 250 millimetre [mm] fork length [FL]) are estimated to be in Snap Lake, and what is the level of confidence in that estimate?"; and, provide a basis for evaluating Lake Trout mortality associated with the fisheries community assessment netting program. A total of 295 Lake Trout were collected from Snap Lake in 2012 through angling, marked with a PIT tag, and released back into the lake. Later, marked fish were recaptured along with unmarked fish. The recaptures for three angling sessions (two in 2012, one in 2013) and one gill netting session (in 2013) were used to derive multiple population estimates based on a median survival rate of 72.2 percent (%) (lower credibility limit 63.0%, upper credibility limit 80.5%). The median estimate of 1,589 fishable Lake Trout in Snap Lake (lower credibility limit = 1,151, upper credibility limit = 2,299) in 2013 was robust as it showed good agreement with estimates made using a different survival rate (i.e., 90% instead of 72.2%) over a shorter time period (i.e., one year instead of two years) or using different sampling gear (gill netting instead of angling). The abundance of Lake Trout in Snap Lake on a unit area basis (e.g., Lake Trout per hectare) was lower than reported for other lakes in the published literature and may be related to the limited amount of suitable Lake Trout habitat available during summer.

Stable Isotope Food Web Analysis Special Study

A stable isotope study was conducted in Snap Lake to answer two key questions: what eats what in Snap Lake; and, is the Snap Lake food web planktonically or benthically driven. Measures of the stable isotope ratios ^{13}C:^{12}C and ^{15}N:^{14}N in tissues of pelagic organisms (zooplankton), profundal benthic organisms (chironomids, oligochates, fingernail clams), littoral benthos (snails, Ephemeroptera, caddisflies), periphyton, and five species of fish (Lake Trout, Round Whitefish, Longnose Sucker, Lake Chub, and Burbot) were evaluated to estimate the diets of each of the five fish species within Snap Lake. Both Lake Trout and Burbot were generalists consuming both fish and invertebrates (profundal, littoral, and pelagic). Lake Trout were the top predator based on trophic position. Round Whitefish, Longnose Sucker, and Lake Chub consumed mixtures of pelagic, profundal, and littoral organisms. The Snap Lake food web was benthically driven in 2013 with 75 percent of the carbon estimated to be derived from benthic sources. Based on enrichment of δ^{13}C but no change for δ^{15}N in Lake Trout between 1999 and 2013, the trophic structure of the Snap Lake food web has been maintained, and while Lake Trout appear to be more reliant on benthos in 2013 than in 1999 the cause or causes are unknown.

Weight of Evidence

The Weight of Evidence (WOE) integration section of the 2013 AEMP combines the information and conclusions of the water quality, fish tissue chemistry, plankton (small animals and plants living in the lake waters), benthic invertebrate community (small animals without backbones living in the lake sediments), and fish community monitoring sections. Sediment quality information was also incorporated from the 2012 AEMP. A qualitative process was used to estimate the strength (or weight) of evidence for nutrient enrichment or toxicological impairment in Snap Lake.

Nutrient enrichment refers to the process whereby nutrients such as nitrates and phosphorus in effluent released to Snap Lake stimulate growth of phytoplankton (small plants) at the base of the food chain. Although beneficial in small amounts, excessive nutrients could have negative impacts on the lake’s existing biological community. Toxicological impairment refers to the process whereby substances such as metals released to the lake can cause toxicity, for instance, reduced growth, reproduction, or survival of the plants and animals in the lake. The integration process combined laboratory determinations of nutrient (chemicals that may cause enrichment) and toxicant exposure (chemicals that may cause toxic effects) with measurements of field biological responses in the plankton, benthic invertebrates and fish. The strength of evidence for either nutrient enrichment or toxicological impairment in Snap Lake was characterized.

For 2013 there appeared to be a clear link between nutrient releases to Snap Lake as a result of Mine activities, stimulation of phytoplankton, and a resulting moderate-level shift in the phytoplankton community. There was also evidence of this nutrient enrichment transferring through the food chain (i.e., as increased food supply) to benthic invertebrates with higher densities of some dominant taxa in Snap Lake. There was no evidence of enrichment transferring to the fish community. In contrast, there was also evidence, albeit weaker, of possible toxicological impairment of zooplankton (small animals without backbones living in the lake waters) and benthic invertebrates (lower density of one taxonomic group), resulting from increases in the concentrations of some substances in water and sediment. The evidence for toxicological impairment was considered uncertain because the observed responses were very mild and could also have been caused by increased predation (fish eating higher numbers of zooplankton and benthic invertebrates) or a change in food supply (phytoplankton). There was no evidence of adverse effects to the structure and function of the Snap Lake ecosystem.

Concentrations of some metals in Lake Trout and Round Whitefish muscle were higher in Snap Lake in 2013 compared to either reference lakes or baseline (i.e., pre-Mine) conditions. The three most consistent changes were observed for cesium, strontium, and thallium, of which only strontium has shown corresponding increases in water quality and sediment quality in Snap Lake. There was no evidence of a Mine-related toxicity response in the fish community suggesting that the increased metals exposure has not caused any impairment to Lake Trout and Round Whitefish.

Action Levels

The Snap Lake Aquatic Effects Monitoring Program (AEMP) Response Framework provides a step-by-step approach for responding to the results of the AEMP. If something of concern is identified (a trigger) that suggest a change or a pattern is not acceptable, De Beers Canada Inc. (De Beers) must take action (a response to the monitoring result). The specific responses will depend on the type and seriousness of any effect(s). In the 2013 AEMP, some items were found to be increasing: cesium and thallium in fish tissue and chloride, fluoride, and nitrate in water. De Beers will be required to submit a plan to address the fish tissue changes. De Beers submitted a Water Licence amendment request for the water quality changes, specifically related to total dissolved solids and its constituent ions, to the Mackenzie Valley Land and Water Board in December 2013.

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MASTER GLOSSARY

^{15}N	A rare stable isotope of nitrogen used to estimate trophic position relative to a baseline value.
a posteriori	After the fact; without prior knowledge. A posteriori weighting criteria are derived after data have been collected.
a priori	Derived by reasoning in advance. A priori weighting criteria are established before data have been collected.
absolute abundance	A measure of change in animal population over time based on changes in the number of animals present.
acidification	The decrease of acid neutralizing capacity in water, or base saturation in soil, caused by natural or anthropogenic processes. Acidification is exhibited as the lowering of pH.
Action Level	A categorization indicating the severity of possible effects to an assessment endpoint in the AEMP
acute	A stimulus severe enough to rapidly induce an effect; in aquatic toxicity tests, an effect observed in 96 hours or less is typically considered acute. When referring to aquatic toxicology or human health, an acute effect is not always measured in terms of lethality.
alkalinity	A measure of water's capacity to neutralize an acid. It indicates the presence of carbonates, bicarbonates and hydroxides, and less significantly, borates, silicates, phosphates, and organic substances. Alkalinity is expressed as an equivalent of calcium carbonate. Its composition is affected by pH, mineral composition, temperature, and ionic strength. However, alkalinity is normally interpreted as a function of carbonates, bicarbonates, and hydroxides. The sum of these three components is called total alkalinity.
allochthonous	Sources of carbon and nutrients that originate outside of an aquatic system (i.e., from terrestrial sources).
age frequency	A means of defining the structure of a population by assigning all individuals to age groups.
annulus	A ring-shaped region found on fish ageing structures (i.e., scales, cleithra, otoliths, fin rays) that represents one year of growth.
area-weighted catch-per-unit-effort	Catch-per-unit-effort values calculated proportionally for predefined habitat strata (i.e., depth ranges) summed together.
assimilation	The process of absorbing nutrients from the diet and converting it to body mass.

asymptote	A point in a curvilinear relationship between two correlated variables beyond which, for an increase in one variable there is no further increase in the other variable.
autochthonous	Sources of carbon and nutrients that originate within an aquatic system.
autotroph	An organism that produces complex organic compounds (such as carbohydrates, fats, and proteins) from simple inorganic molecules using energy from light (by photosynthesis) or inorganic chemical reactions (chemosynthesis). They are the producers in a food chain, such as plants on land or algae in water.
background	An area not influenced by chemicals released from the site under evaluation.
baseline	A surveyed or predicted condition that serves as a reference point to which later surveys are coordinated or correlated.
bathymetry	Measurement of the depth of a waterbody.
Bayesian analysis	A statistical method of inference that derives the posterior probability as a consequence of two antecedents, a prior probability, and a "likelihood function" derived from a probability model for the data to be observed.
benchmark	A standard or point of reference against which things may be compared or assessed.
benthic invertebrates	Invertebrate organisms living at, in, or in association with the bottom (benthic) substrate of waterbodies such as lakes, ponds, and streams. Examples of benthic invertebrates include some aquatic insect species, such as caddisfly larvae, that spend at least part of their life stages dwelling on bottom sediments in the waterbody. These organisms play several important roles in the aquatic community. They are involved in the mineralization and recycling of organic matter produced in the water above, or brought in from external sources, and they are important second and third links in the trophic sequence of aquatic communities. Many benthic invertebrates are major food sources for fish.
bioaccumulate	The accumulation of substances in an organism
biochemical oxygen demand	An empirical test in which standardized laboratory procedures are used to determine the relative oxygen requirements of wastewaters, effluents, and contaminated waters.
biomagnify	The increase in concentration of a substance in organisms that increases with each increase in trophic level throughout a food chain.
biomass	The weight or mass of a particular species or taxa usually in a specific unit of area.
biota	Living organisms.
biotelemetry	A method of attaching a device to an organism that provides information about the animal to which the device is attached.

Boreal Forest	The northern hemisphere, circumpolar, tundra forest type consisting primarily of black spruce and white spruce with balsam fir, birch, and aspen.
Broad-scale Community Monitoring (BsM)	A standardized method of gill-netting using "small" and "large" mesh gill-nets set randomly in a depth stratified fashion to estimate the relative abundance of fish populations within a lake or reservoir.
calcareous	Refers to a type of shell that largely comprised of calcium carbonate.
Canadian Water Quality Guideline (CWQG) for the Protection of Aquatic Life	Guidelines established by the Canadian Council of Ministers of the Environment and used to assess the potential effects of the concentration of different water quality parameters upon aquatic life (i.e., fish, aquatic plants [macrophytes], and benthic invertebrates). Exceedance of a guideline does not mean that adverse effects will occur with certainty, only that they may occur and that this possibility needs to be investigated further.
capture probability	The probability that an animal will be captured during a session. In closed population models, capture probability refers only to unmarked animals.
catch curve	A descriptive figure that describes catch-per-unit-effort for discrete age groups.
catch-per-unit-effort	The number of individuals or biomass caught per unit of time, area, or distance with a specific type of gear.
chlorophyll *a*	The primary photosynthetic pigment contained in the phytoplankton (primary producers).
Chlorophyta	Green algae; a component of phytoplankton.
chronic	The development of adverse effects after extended exposure to a given substance. In chronic toxicity tests, the measurement of a chronic effect can be reduced growth, reduced reproduction or other non-lethal effects, in addition to lethality. Chronic should be considered a relative term depending on the life span of the organism.
Chrysophyta	Golden-brown algae; a component of phytoplankton.
cladocera	A group of small planktonic animals (crustaceans) also known as water fleas; a component of zooplankton.
cleithrum	A membrane bone which first appears as part of the skeleton in primitive bony fish, where it runs vertically along the scapula.
closed population	A group of individuals in a particular area studied over a defined time period when there is no birth, death, immigration, and emigration.
cloth holding cradle	A special device for landing long fish composed of two long poles attached together by an equally long piece of cloth.
colonial	Individuals of the same species clustered together to form a group.

community composition	The species present and their relative proportions in an assemblage of organisms.
conductivity	A measure of the capacity of water to conduct an electrical current. It is the reciprocal of resistance. This measurement provides an estimate of the total concentration of dissolved ions in the water.
continuous flow	Refers to a type of automated sample preparation device and mass spectrometer that uses a continuous stream of helium carrier gas.
copepoda	An order of planktonic crustaceans; a component of zooplankton.
covariate	In statistics, a covariate is a variable that is possibly predictive of the outcome under study.
credibility limits	An interval in the domain of a posterior probability distribution used for interval estimation in Bayesian statistics.
critical effect size	A threshold above which an effect may be indicative of a higher risk to the environment (Metal Mining Environmental Effects Monitoring Guidance Document; Environment Canada 2012).
Cryptophyta	Flagellated algae also known as cryptomonads; a component of phytoplankton.
Cyclopoida	An order of copepods; small planktonic animals.
detection limit (DL)	The lowest concentration at which individual measurement results for a specific analyte are statistically different from a blank (that may be zero) with a specified confidence level for a given method and representative matrix.
dewatering	Removal of water; e.g., removal of groundwater from surficial aquifers or deposits using wells or drainage ditch systems; removal of water from lakes to allow mining.
diatom	A group of algae that are encased within a frustule (a shell) made of silica; a component of phytoplankton.
diffuser	A device used to disperse an effluent plume to a waterbody.
diffuser ports	Holes at the end of a diffuser where effluent is discharged.
diffuser station	Monitoring station located less than 200 metres from the diffuser.
discrimination	Refers processes (i.e., uptake, assimilation and excretion) that affect the relative abundance of stable isotopes in an organism's tissues.
discrimination factor	The difference in isotopic composition of a consumer and its diet.
dissolved oxygen (DO)	Measurement of the concentration of dissolved (gaseous) oxygen in the water, usually expressed in milligrams per litre (mg/L).

duplicate field sample	A second sample collected at the same time and from the same location, repeating the same collection procedure as the original sample. Such a sample is used to detect variability at a site and verify the field-sampling method.
duplicate laboratory sample	A water sample that is submitted to the laboratory is split into two samples by the analytical laboratory, each tested separately. These samples are used to assess the reproducibility of the laboratory results (i.e., laboratory method and analyses).
ecosystem	An integrated and stable association of living and non-living resources functioning within a defined physical location. A community of organisms and its environment functioning as an ecological unit. For the purposes of assessment, the ecosystem must be defined according to a particular unit and scale.
effluent	Stream of water discharging from a source.
Ekman grab	Cube-shaped mechanical device with a spring-loaded opening that is lowered to the bottom of a waterbody and triggered to close to collect a sample of the bottom substrate.
electrofishing	A live fish capture technique in which negative (anode) and positive (cathode) electrodes are placed in the water and an electrical current is passed between the electrodes. Fish are attracted to the negative electrode and become stunned by the current, allowing fish to be collected, measured and then released.
elutriate	To purify or separate by washing and straining.
embayment	A bay or protected area in a waterbody such as a lake.
epaxial muscle	The muscles of the upper (i.e., dorsal) half of the body in fish.
epilimnion	The uppermost and in summer warmest water layer in a lake or other standing water body when the water body is thermally stratified.
epilithic	Aquatic plants that grow on the surface of rocks.
euphotic	The upper surface layer of a waterbody where sufficient light penetrates to allow photosynthesis to occur.
eutrophication	The over-fertilization of a body of water, which generally results in increased plant growth and decay. This ultimately leads to an increase in simple algae and plankton over more complex plant species, resulting in a decrease in water quality. Causes of eutrophication can be anthropogenic or natural.
exploited	A population of animals that is subject to harvest.
exposure lake	A lake potentially impacted by an external disturbance.
far-field	Stations located in the southern portion of the south basin of Snap Lake, and in the northeast and southeast arm of Snap Lake.
field blank	A solution of de-ionized water provided by the laboratory that is used to detect sample contamination during the collection, shipping, and analyses of samples.

field specific conductivity	A measurement of how well water conducts electricity, from a conductivity meter used on site.
filamentous	A long chain of cells.
fin rays	The hornlike, cartilaginous, or bony, dermal rods which form the skeleton of the fins of fishes.
fish	Fish as defined in the *Fisheries Act*, includes parts of fish, shellfish, crustaceans, marine animals, and any parts of shellfish, crustaceans or marine animals, and the eggs, sperm, spawn, larvae, spat, juvenile, and adult stages of fish, shellfish, crustaceans, and marine animals.
fish condition	The relative plumpness of a fish; various indices are used to quantify fish body condition.
fishable population	The portion of the fish community susceptible to a specific method of fish capture.
Fisheries and Oceans Canada (DFO)	Responsible for policies and programs in support of Canada's economic, ecological, and scientific interests in oceans and inland waters; for the conservation and sustainable utilization of Canada's fisheries resources in marine and inland waters; for leading and facilitating federal policies and programs on oceans; and, for safe, effective, and environmentally sound marine services responsive to the needs of Canadians in a global economy.
Fulton's K	A measure of an individual fish's health or plumpness that uses standard weight and length.
Generalist	An organism that is able to utilize a variety of habitats and food resources.
geographic information system (GIS)	Computer software designed to develop, manage, analyze, and display spatially referenced data.
girth	The measurement around something; circumference.
global positioning system (GPS)	A system of satellites, computers, and receivers that is able to determine the latitude and longitude of a receiver on Earth by calculating the time difference for signals from different satellites to reach the receiver.
grab water sample	A single discrete water sample that is collected from a waterbody.
groundwater	That part of the subsurface water that occurs beneath the water table, in soils and geologic formations that are fully saturated.
habitat	The place or environment where a plant or animal naturally or normally lives or occurs.

headwater	The source and upper reaches of a stream or reservoir. The water upstream from a structure or point on a stream. The small streams that come together to form a river. Also may be thought of as any and all parts of a river basin except the mainstem river and main tributaries.
herbivory	A mode of feeding in which an organism known as a herbivore consumes only autotrophs such as plants, algae, and photosynthesizing bacteria.
heterogeneity	Consisting of parts that are unlike each other. For example, the variety and abundance of ecological units (e.g., different terrestrial and water ecosystems) comprising a landscape mosaic.
histology	The microscopic study of tissues.
homogeneity	The quality of being similar or comparable in kind or nature.
homoscedasticity	In statistics, a sequence or a vector of random variables is homoscedastic if all random variables in the sequence or vector have the same finite variance. This is also known as homogeneity of variance.
hydrology	The science of water movement and distribution, including the hydrologic cycle and interactions with the physical and biological environment.
hypolimnion	The lowermost and in summer the coldest water layer in a lake or other standing water body that is thermally stratified.
ice-covered conditions	The period of time, during the year, when waterbodies are covered in ice.
interaction term	In statistics, an interaction is a term in a statistical model in which the effect of two, or more, variables is not simply additive.
isotherm	A certain temperature layer across the area of a lake having the same temperature.
isotope	Atoms of the same element that have the same number of protons in their nuclei, but a different number of neutrons (e.g., ^{12}C, ^{13}C and ^{14}C).
Isotopic mixing model	A tool for inferring the relative contribution of different source items (i.e., prey) to a mixture (i.e., consumer) based on stable isotope ratios in prey and consumer tissues.
jigging	An active form of angling in which the hook is moved in a vertical fashion at varying depths.
juvenile fish	Fish that are no longer young-of-the-year but that have not yet reached reproductive maturity.
kimberlite	Igneous rocks (i.e., formed by the solidification of molten lava) that originate deep in the mantle and intrude the Earth’s crust. These rocks typically form narrow pipe-like deposits that sometimes contain diamonds.
labile	Susceptible to alteration or destruction.

laboratory specific conductivity	A measurement of how well water conducts electricity, as measured in the laboratory.
length at infinity	This is the length (Linf) that the fish of a population would reach if they were to grow indefinitely.
life history	The series of changes undergone by an organism during its lifetime.
littoral zone	The zone in a lake that is closest to the shore. It includes the part of the lake bottom and its overlying water, between the highest water level and the depth where there is enough light (about 1% of the surface light) for rooted aquatic plants and algae to colonize the bottom sediments.
Lugol's solution	Can be used to test for the presence of starch.
main basin	The main basin of Snap Lake excluding the northwest arm.
mark recapture	A method commonly used in ecology to estimate an animal's population's size. Where a portion of the population (N) is tagged (M) and released and the number of fish recaptured with a tag (m) in a recapture sample (n) used to estimate population size (e.g., N=Mn/m).
mesotrophic	Trophic state classification for lakes characterized by moderate productivity and nutrient inputs (particularly total phosphorus).
metabolic power capacity	Energy flow in watts available per unit time for a fish to support critical daily life support activities.
metalimnion	Zone of rapid temperature change within the water column.
method blank	A laboratory grade, pure water sample that is subjected to all laboratory procedures. Used to detect the possibility of cross-contamination between samples in the laboratory.
method detection limit (MDL)	The minimum concentration of a substance that can be measured and reported with a 99% level of confidence.
microcystin	Toxic substance produced by cyanobacteria.
microcystin-LR	The most toxic microcystin.
mid-field	Stations located in the northern half of the south basin of Snap Lake.
mixing zone	The region in which the initial dilution of a discharge occurs.
morphology	The study of the forms of things, both living and non-living (e.g., how erosion affects shape).
mortality rate	Mortality rate is a measure of the number of deaths (in general, or due to a specific cause) in a population, scaled to the size of that population, per unit of time.
near-field	Stations located in the north basin of Snap Lake.

normal range	An estimate of natural variability.
normality	The act of determining if a data set is well-modeled by a normal distribution.
northwest arm (NW arm)	The arm of Snap Lake located north and west of the Mine.
nutrients	Environmental substances (elements or compounds) such as nitrogen or phosphorus, which are necessary for the growth and development of plants and animals.
oligo-mesotrophic	A lake with low to moderate concentration of nutrients and low to moderate organic productivity.
oligotrophic	Trophic state classification for lakes characterized by low productivity and low nutrient inputs (particularly total phosphorus).
ontogenetic change	Refers to a change in diet associated with an increase in body size.
open-water conditions	The period of time during the year when waterbodies are relatively free of ice.
open-water season	Same as above.
outlier	A data point that falls outside of the statistical distribution defined by the mean and standard deviation.
***P*–value**	Statistical value used to determine the significance of a relationship or difference, e.g., P <0.05.
particulate matter	A mixture of small particles, e.g., dust and soil.
Pee Dee belemnite	A Cretaceous marine fossil from the Pee Dee formation. The fossil is used as the internationally-accepted standard for expression of carbon and oxygen stable isotope measurements.
pelagic	Open-water area within a lake.
periphyton	A mixture of microbes, algae, bacteria and plant detritus that is found on submerged surfaces.
Peterson method	A simple mark-recapture technique used to estimate abundance of animals. Usually has one marking and one recapture event.
piscivore	A fish that feeds on fish.
pH	The degree of acidity (or alkalinity) of soil or solution. The pH scale is generally presented from 1 (most acidic) to 14 (most alkaline). A difference of one pH unit represents a ten-fold change in hydrogen ion concentration.

plankton	Small, often microscopic, plants (phytoplankton) and animals (zooplankton) that live in the open-water column of non-flowing water bodies such as lakes. They are an important food source for many larger animals.
plume	The form effluent takes in water following discharge.
pollution	Contamination that results in adverse biological effects to populations or communities of organisms.
polygon	Representations of an area consisting of a plane figure bounded by straight edges.
population	A group of individuals in a defined area.
prediction interval	Is an estimate of an interval in which future observations will fall, with a certain probability, given what has already been observed.
primary productivity	The rate of primary production or the production of organic compounds through the process of photosynthesis or chemosynthesis.
probability distribution	A distribution that assigns a probability to each measurable subset of the possible outcomes of a random experiment.
probable effect levels	Concentration of a chemical in sediment above which adverse effects on an aquatic organism are likely but not certain to occur.
profundal	The deep zone of a waterbody that is below the range of light penetration.
pseudocoelomate	Any of a group of invertebrates with a three-layered body that has a fluid-filled body cavity (pseudocoelom) between the innermost and middle tissue layers.
pseudoreplication	Occurs when, due to experimental design, sampling, or statistical analyses, related samples are treated as independent replicates.
quality assurance (QA)	Management and technical practices designed so that the data generated are of consistent high quality. They include standardization and review by field and office personnel of procedures used in the collection, transport, and analyses of samples.
quality control (QC)	Internal techniques used to measure and assess data quality, including samples that are used to detect and reduce systematic and random errors that may occur during field sampling and laboratory procedures.
R^2	A coefficient of determination, a statistical measure of how well a regression line approximates the real data points.
recruitment	A term in fisheries science denoting the number of individuals in a population that hatch in a year and survive to reproductive size.
reference lake	A sampling lake selected for its relatively undisturbed conditions.
relative abundance	The proportional representation of a species in a sample or a community.

relative weight index	An index of condition in fish in which the actual weight of a fish is divided by a standard weight selected to represent an optimal weight for that species at that length which is then multiplied by 100.
reproductive potential	The relative capacity of a species to reproduce itself under optimum conditions.
rotifer	A large class of the phylum Aschelminthes; a component of zooplankton.
Secchi depth	A measure of water clarity, measured by lowering a 20 cm diameter disk (Secchi disk) with alternating black and white coloured quadrants. The shallowest depth at which the disk is no longer visible is the Secchi depth. High secchi depth readings indicate clearer water that allows sunlight to penetrate to greater depths. Low readings indicate turbid water which can reduce the passage of sunlight to bottom depths. Limited light penetration can be a factor in diminished aquatic plant growth beneath the surface, thus reducing the biological re-aeration at lower depths.
secondary productivity	The rate at which an ecosystem's consumers convert the chemical energy of the food they eat into their own new biomass.
sediment	Solid material that is transported by, suspended in, or deposited from water. It originates mostly from disintegrated rocks; it also includes chemical and biochemical precipitates and decomposed organic material.
sedimentation	The process of subsidence and deposition of suspended matter carried by water, wastewater or other liquids, by gravity. It is usually accomplished by reducing the velocity of the liquid below the point at which it can transport the suspended material.
senescence	The aging process in mature individuals; the period near the end of an organism's life cycle.
sentinel species	Species that can be used as an indicator of environmental conditions.
Simpson’s diversity index	Used to measure diversity. In ecology, it is often used to quantify the biodiversity of a habitat. It takes into account the number of species present, as well as the relative abundance of each species. The Simpson index represents the probability that two randomly selected individuals in the habitat will not belong to the same species.
specialist	An organism that is able to utilize only a narrow range of habitats and food resources.
specific conductivity	A measure of how well water conducts electricity.
spring freshet	A spring thaw event resulting from melting snow and ice.

stable isotope	A form of the same element containing equal numbers of protons but different numbers of neutrons. In ecology the ratios of ^{13}C and ^{12}C, and ^{15}N and ^{14}N in organisms are used to infer the sources of carbon and the trophic level of the organism; not radioactive, they do not spontaneously undergo radioactive decay.
standard deviation (SD)	A measure of the variability or spread of the measurements about the mean. It is calculated as the positive square root of the variance.
standard error (SE)	The standard deviation (positive square-root of the variation) of the errors associated with a series of measurements.
stenotherm	A species only capable of living or surviving within a narrow temperature range.
stratify	Layering of lakes into two or more non-mixing layers; in summer, typically a layer of warmer, less dense water lies on a cooler, denser layer; in winter, typically a layer of very cold (<4°C), less dense water overlies warmer, denser water (approximately 4°C).
stressors	Physical, chemical, or biological perturbations to a system that are either (a) foreign to that system or (b) natural to the system but applied at an excessive [or deficient] level. Stressors cause significant changes in the ecological components, patterns and processes in natural systems. Examples include water withdrawal, pesticide use, timber harvesting, traffic emissions, stream acidification, trampling, poaching, land-use change and water pollution.
taxa	A group of organisms of any taxonomic rank (e.g., family, genus, species).
taxon	A group of organisms at the same level of the standard biological classification system; the plural of taxon is taxa.
thermal suitability	Areas that meet the thermal requirements of stenotherms.
thermocline	The depth zone at which water temperature declines by more than 1°C for every meter of increasing depth during periods of thermal stratification of lakes and reservoirs.
thermoregulate	The ability of an organism to keep its body temperature within certain boundaries, even when the surrounding temperature is very different and sometimes accomplished by moving to a particular temperature zone in a lake (i.e., behavioural thermoregulation).
tissue metal concentrations	The concentration of heavy metals within various tissues (i.e. kidney, liver, and muscle) of fish.
total dissolved solids (TDS)	The total concentration of all dissolved solids found in a water sample.
total Kjeldahl nitrogen (TKN)	The sum of organic nitrogen, ammonia, and ammonium.

total organic carbon (TOC)	Composed of both dissolved and particulate forms; often calculated as the difference between total carbon and total inorganic carbon. Total organic carbon has a direct relationship with both biochemical and chemical oxygen demands, and varies with the composition of organic matter present in the water. Organic matter in soils, aquatic vegetation, and aquatic organisms are major sources of organic carbon.
total suspended solids (TSS)	The amount of suspended substances in a water sample. Solids, found in water, which can be removed by filtration.
toxicity	The inherent potential or capacity of a material to cause adverse effects to a living organism.
Traditional Knowledge	Knowledge and understanding of traditional resource and land use, harvesting, and special places.
travel blank	A water sample prepared by the laboratory and shipped to the field sampling location and subsequently returned to the laboratory unaltered. These samples are used to detect sample contamination during transport.
trophic	Pertaining to part of a food chain, for example, the primary producers are a trophic level just as tertiary consumers are another trophic level.
trophic level	A functional classification of organisms in an ecosystem according to feeding relationships, from primary producers through herbivores (primary consumers) and carnivores (secondary and tertiary consumers).
trophic state	Eutrophication is the process by which lakes are enriched with nutrients, increasing the production of rooted aquatic plants and algae. The extent to which this process has occurred is reflected in a lake's trophic classification or state: oligotrophic (nutrient poor), mesotrophic (moderately productive), or eutrophic (very productive and fertile).
t-test	Statistical test used to compare between two groups of samples.
turbidity	An indirect measure of suspended particles, such as silt, clay, organic matter, plankton, and microscopic organisms, in water.
turnover	Refers to the replacement of carbon and nitrogen in tissues, and a subsequent change in stable isotope composition of tissues, following a change in diet.
under ice	The period of year when the lakes are partially or completely covered with ice.
utildor	An enclosed insulated conduit running above ground that is used to carry water, sewage or electricity between buildings constructed on permafrost.
vertical mixing	The mixing of different substances through the water column to yield homogeneous concentrations of different parameters throughout a lake.
vertical profile	An *in situ* measurement consisting of taking readings of physical parameters or samples at certain depth increments in the water column of a lake.

von Bertalanffy growth model	A growth model that predicts the length of a fish as a function of its age.
waterbody	Any location where water flows or is present, whether or not the flow or presence of water is continuous seasonal, intermittent, or occurs only during a flood.
watercourse	Riverine systems such as creeks, brooks, streams, and rivers.
watershed	The entire catchment area of runoff containing a single outlet.
weight-of-evidence	A process used in ecological risk assessments and environmental monitoring by which multiple measurement endpoints (often referred to in this context as "lines of evidence") are related to an assessment endpoint for a particular receptor.
wetlands	Wetlands are land where the water table is at, near or above the surface or which is saturated for a long enough period of time to promote such features as wet-altered soils and water tolerant vegetation. Wetlands include organic wetlands or "peatlands," and mineral wetlands or mineral soil areas that are influenced by excess water but produce little or no peat.
young-of-the-year	Fish at age 0, within the first year after hatching.
YSI	A meter that measures temperature, conductivity, and dissolved oxygen in water.
zooplankton	Small, sometimes microscopic, animals that live in the water column of non-flowing waterbodies such as lakes and mainly eat primary producers (phytoplankton).

SECTION 1

INTRODUCTION

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LIST OF ACRONYMS

Term	Definition
AANDC	Aboriginal Affairs and Northern Development Canada
AEMP	Aquatic Effects Monitoring Program
De Beers	De Beers Canada Inc.
EAR	Environmental Assessment Report
Golder	Golder Associates Ltd.
HDPE	high density polyethylene
Mine	Snap Lake Mine
MVEIRB	Mackenzie Valley Environmental Impact Review Board
MVLWB	Mackenzie Valley Land and Water Board
MVRB	Mackenzie Valley Review Board
NWT	Northwest Territories
SNP	Surveillence Network Program
WLWB	Wek´èezhìi Land and Water Board
WOE	Weight of Evidence

UNITS OF MEASURE

Term	Definition
km	kilometre

1 INTRODUCTION

1.1 Background

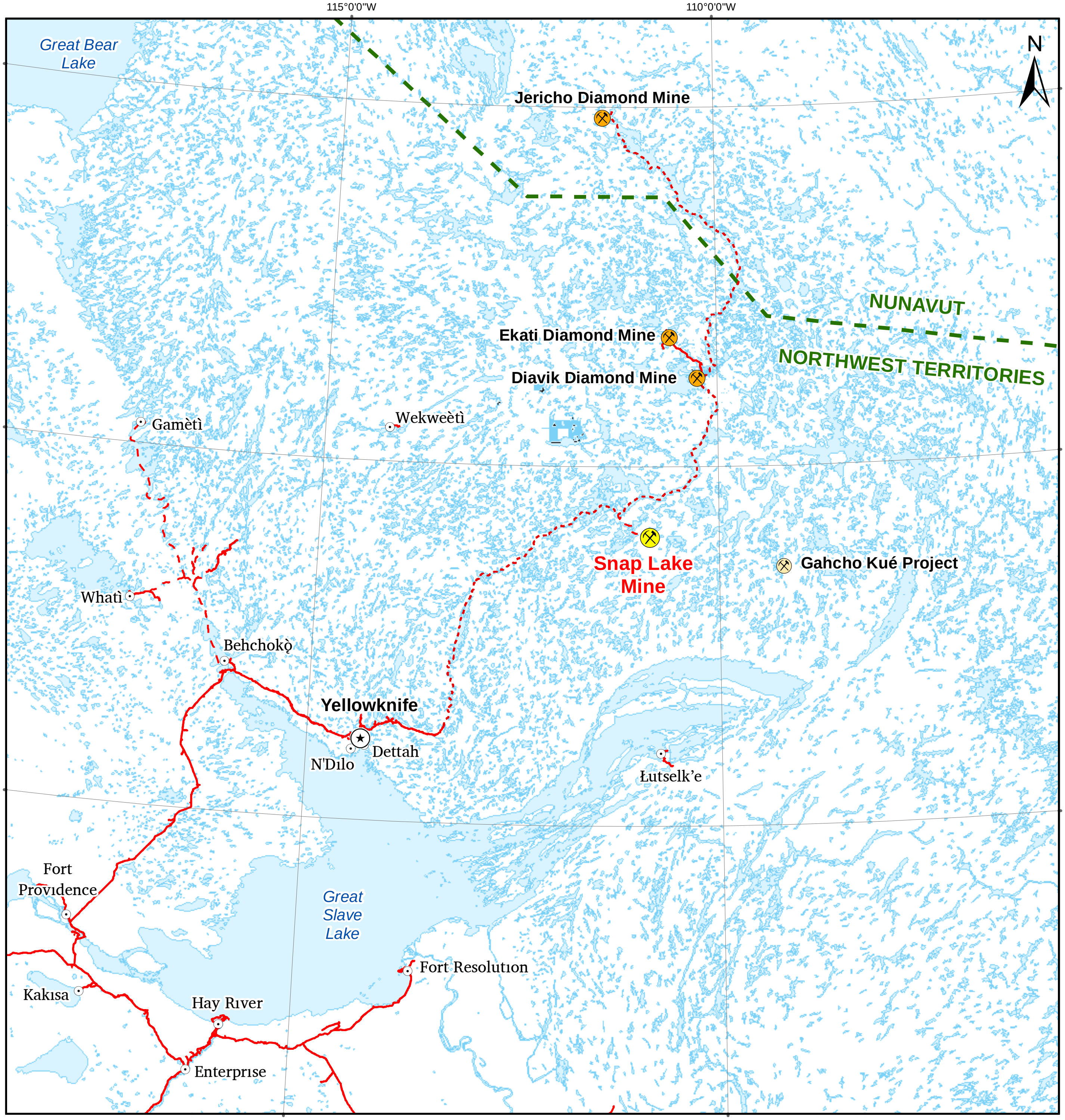
De Beers Canada Inc. (De Beers) owns and operates the Snap Lake Mine (the Mine), a diamond mine located approximately 220 kilometres (km) northeast of Yellowknife, Northwest Territories. The Mine is located 30 km south of MacKay Lake and 100 km south of Lac de Gras, where the Diavik and Ekati diamond mines are located (Figure 1-1).

An Environmental Assessment Report (EAR) for the Mine (De Beers 2002) was submitted to the Mackenzie Valley Review Board (MVRB) in February 2002. The Mine received approval from the Minister of Indian and Northern Affairs (now Aboriginal Affairs and Northern Development Canada [AANDC]) in October 2003, based on a decision report and recommendation from the MVRB (MVEIRB 2003). In 2004, De Beers negotiated an Environmental Agreement and received the required Water Licence, Land Use Permit, Land Leases, and *Fisheries Act* Authorization to begin construction and operation of the Mine.

In June 2011, the Mine submitted an application to renew the Water Licence, and hearings were subsequently held in December 2011. The Water Licence was renewed for a period of eight years, effective June 14, 2012, and was amended in 2013 (Water Licence MV2001L2-0004; MVLWB 2013a).

The Aquatic Effects Monitoring Program (AEMP) is a requirement of the Water Licence Part G (MVLWB 2013a). The goal of the AEMP is to address potential Mine-related effects to the aquatic ecosystem of Snap Lake in a scientifically defensible manner. The first AEMP Design Plan was submitted in 2004; the scope of the AEMP from 2005 to 2012 was based on the 2005 AEMP Design Plan submitted to the Mackenzie Valley Land and Water Board (MVLWB) in June 2005. The scope of the 2013 AEMP was based on the final 2013 AEMP Design Plan (De Beers 2014).

As stated in Part G Item 3 of the current Water Licence, De Beers is to submit an update to the AEMP Design Plan in 2012 and every four years thereafter for MVLWB approval. The intent of updating the AEMP Design Plan is to provide De Beers the opportunity to make modifications according to the findings of the previous year of monitoring. The Draft 2013 AEMP Design Plan was submitted to the MVLWB in November 2012. The monitoring portion of the Draft 2013 AEMP Design Plan was approved by the MVLWB on March 28, 2013 (MVLWB 2013b). At that time, the MVLWB requested revisions to Sections 6 and 7 (Weight of Evidence [WOE] and Response Framework). On November 29, 2013, the MVLWB approved sections 6 and 7 of the 2013 AEMP Design Plan (MVLWB 2013c). The final 2013 AEMP Design Plan was submitted to the MVLWB in January 2014 (De Beers 2014).



I:\CLIENTS\DE_BEERS\Snap_Lake\Mapping\MXD\AEMP2013\Figure_1_1_Location_of_SnapLake_FINAL.mxd

LEGEND

Snap Lake Mine	Watercourse
Existing Mine	Waterbody
Proposed Mine	
Territorial Capital	
Populated Place	
All Weather Road	
Tibbit-to-Contwoyto Winter Road	
Winter Road	
Territorial/Provincial Boundary	

REFERENCES

Base Data Source: The Atlas of Canada, CanVec

SNAP LAKE MINE

Location of Snap Lake Mine

PROJECTION: UTM Zone 12	DATUM: NAD83	DE BEERS GROUP OF COMPANIES
75 0 75 SCALE 1:3,500,000 KILOMETRES		
FILE No: 1413490004-7001_1_1		DATE: March 27, 2014
JOB NO: 14-1349-0004	REVISION NO: A	Figure 1-1
OFFICE: GOLD-CAL	DRAWN: VV	CHECK: PMC

De Beers used input from traditional knowledge holders provided during the EAR and regulatory process to develop the AEMP. The design of the AEMP and the content of the annual report reflect monitoring priorities identified by northern communities during the design process. Aboriginal community members participated in the fish-tasting event annually (Section 10).

1.2 Objective and Scope

1.2.1 Objective

This document represents the tenth annual AEMP report for the Mine and presents the results of the 2013 AEMP program. The main objectives of the 2013 AEMP Annual report were to describe monitoring results from Snap Lake for Mine-related effects, verify, and update the EAR predictions (De Beers 2002), and provide information to inform management decisions made by the Mine.

An additional objective of the 2013 AEMP Annual Report was to address the requirements specified in Part G, Item 8 of the Water Licence (Table 1-1). Data from relevant Surveillance Network Program (SNP) stations are integrated into the AEMP and are included in this report. All SNP and AEMP monitoring activities are reported in the Water Licence Annual Report.

Table 1-1 AEMP Annual Reporting Requirements Specified in Part G, Item 8 of the Water Licence

	Item	Location in Report
a)	a plain language summary of the major results obtained in the preceding calendar year and a plain language interpretation of the significance of those results;	Executive Summary
b)	a summary of activities conducted under the AEMP;	Section 1.2
c)	an update of the Mine development activities and any accidents, malfunctions or spills within the report time frame that could influence the results of the AEMP;	Sections 1.4 and 2
d)	tabular summaries of all data and information generated under AEMP in a format acceptable to the Board;	Sections to Section 12 and appendices
e)	an interpretation of the results, including an evaluation of any identified environmental effects that occurred as a result of the Mine;	Sections 2 to 12; summarized in Section 12
f)	an analysis that integrates the results of individual monitoring components collected in a calendar year and describes the ecological significance of the results;	Section 12
g)	a comparison of monitoring results to Action Levels as set in the AEMP Design Plan;	Section 13
h)	an evaluation of the overall effectiveness of the AEMP to date,	Section 12
i)	recommendations for refining the AEMP to improve its effectiveness as required; and,	Sections 2 to 11; Section 15
i)	any other information specified in the approved AEMP Design Plan or that may be requested by the Board before November 1 of any year.	Section 11 (Special Studies); Appendix 5B; Appendix 9A

AEMP = Aquatic Effects Monitoring Program; Mine = Snap Lake Mine

1.2.2 Scope

The main component of the AEMP is operational monitoring, which occurs during all phases of the Mine development. The AEMP also allows De Beers to compare Mine-related effects with EAR predictions. The 2013 AEMP components are:

- Site characterization (Section 2);
- Water quality (Section 3);
- Sediment quality (Section 4);
- Plankton (Section 5);
- Benthic invertebrate community (Section 6);
- Fish community (Section 8);
- Fish tissue chemistry (Section 9);
- Fish tasting (Section 10); and,
- WOE (Section 12).

Special studies occur as needed, and include research activities that support effects monitoring. These studies are not part of monitoring activities, as they do not assess changes that may be related to the Mine, but rather focus on development of monitoring methods or further investigation of monitoring findings. Special studies conducted during the 2013 AEMP program and included in this report are: Littoral Zone Special Study (Section 11.1); Picoplankton Special Study (Section 11.2); Downstream Lakes Special Study (Section 11.3); Lake Trout Population Estimate Special Study (Section 11.4); and, Stable Isotope Food Web Analysis Special Study (Section 11.5).

1.2.2.1 Action Levels

The Snap Lake AEMP response framework links monitoring results to Action Levels with the purpose of determining whether assessment endpoints are within an acceptable range (De Beers 2014). The Response Framework includes definitions of significance thresholds and tiered Action Levels applicable to the aquatic environment (Section 13). A significance threshold is a magnitude of environmental change that would result in significant adverse effects (WLWB 2010). An Action Level is a magnitude of environmental change that triggers management action (WLWB 2010).

The goal of the response framework is to systematically respond to monitoring results, as necessary, to identify potential for significant adverse effects and undertake necessary mitigation actions. This is accomplished by implementing appropriate mitigation at predefined Action Levels, which are triggered before a significant adverse effect can occur. Changes from baseline data, reference lake data, or deviations from the range of natural variability are all considered in the determination of whether an Action Level is triggered.

The Action Levels were based on the 2013 AEMP Design Plan (De Beers 2014). This AEMP report represents the first time the Response Framework was applied to the AEMP. Action Levels are discussed in each section, and are summarized in Section 13.

1.3 Study Areas

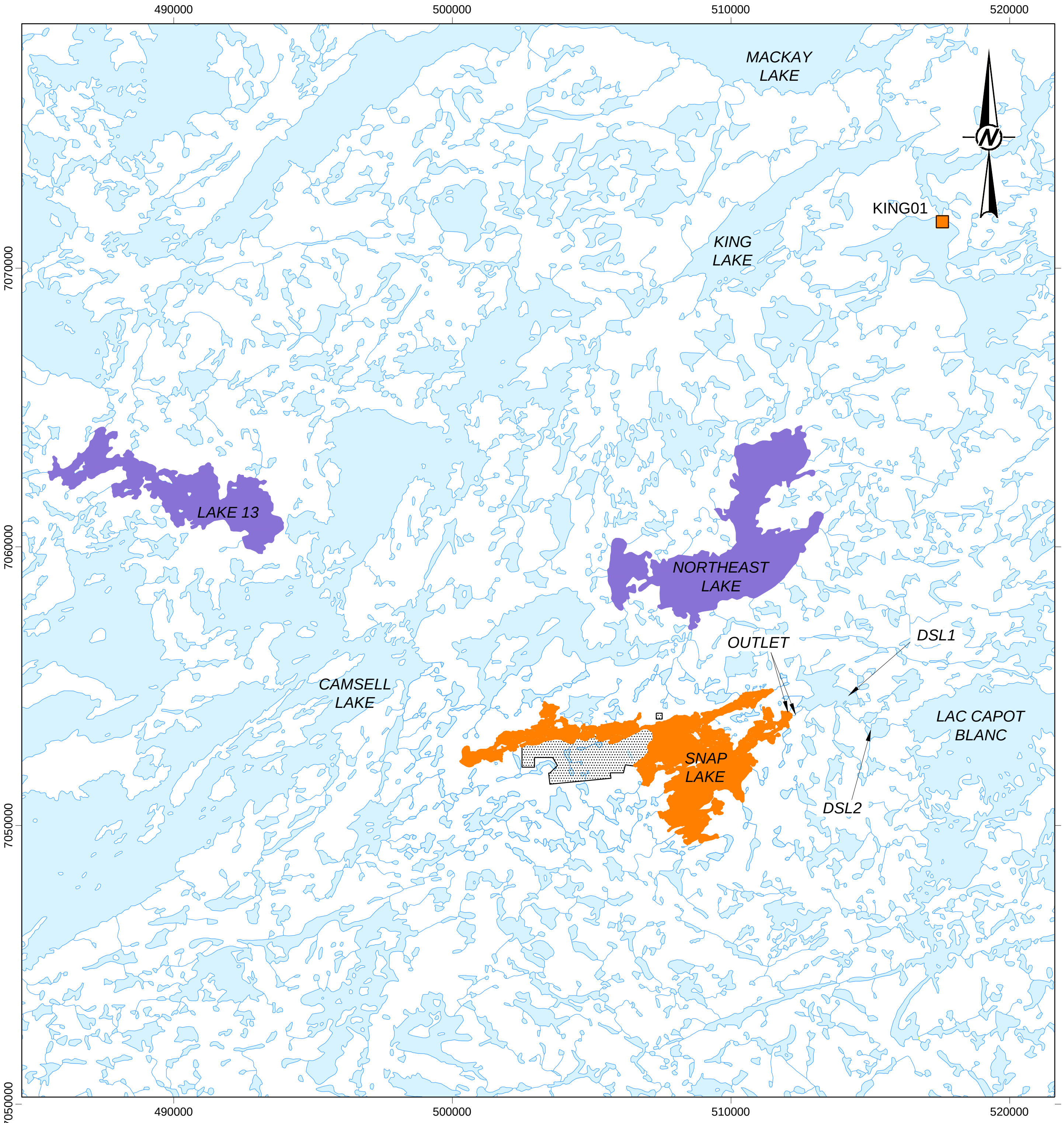
The study areas for the 2013 AEMP monitoring consisted of Snap Lake, Northeast Lake, Lake 13, and one station downstream of Snap Lake in the Lockhart River system, located upstream of King Lake. (Figure 1-2). De Beers also completed a Downstream Lakes Special Study in 2013 that focused on three lakes immediately downstream of Snap Lake (Section 11.3).

1.4 Site Activities In 2013

Major construction activities and milestones achieved during 2013 by De Beers were:

- A 4" high density polyethylene (HDPE) line was installed from fire pumps to the Utilities Building as a dedicated potable supply to prevent contamination events due to scouring of the line. A new interlock valve will divert supply flow when the main fire pumps start, preventing a pulse of contaminated water from reaching the potable water treatment plant intake.
- A process water pipeline was installed in the North Pile to facilitate the pumping of sump water.
- The perimeter sump 3 liner was repaired.
- A new sewage treatment plant was installed in the Utilities Building.
- A modular water treatment plant project was initiated in the Utilities Building.
- A wireless system was installed for monitoring heat trace, pressure, and flow from the new pump shacks in the North Pile.
- A flocculant tank was installed in close proximity to PS3.

As required under Part G, Item 8b of the Water Licence (MV2011L2-0004), De Beers has reviewed site activities for 2013. Spills and leaks that occurred on site during this period were contained and mitigated. The possible influence of spills on water quality in Snap Lake, and their potential effects to organisms living in the lake are considered in this 2013 AEMP report and outlined in further detail in Section 2.



LEGEND

- AEMP Water Quality Station
- Snap Lake Mine Footprint
- Monitoring Area
- Reference Area
- Waterbody
- Watercourse

NOTE

DSL1 First lake downstream of Snap Lake

DSL2 Second lake downstream of Snap Lake

REFERENCE

Digital map from Mackay Lake, Northwest Territories, produced by Department of Energy, Mines and Resources. map 75m, original scale 1:250,000, projection: Transverse Mercator, Datum: NAD83, coordinate system: UTM zone 12.

SNAP LAKE MINE

Study Area, 2013 AEMP

PROJECTION: UTM Zone 12	DATUM: NAD83

4 0 4

SCALE 1:200,000 KILOMETRES

DE BEERS
GROUP OF COMPANIES

FILE No: 1313490001-7010_1-2		DATE: March 26, 2014
JOB No: 14-1349-0004	REVISION No: A	
OFFICE: GOLDER VICTORIA	DRAWN: JEF	CHECK: PMC

Figure **1-2**

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1.5 2013 Report Organization

The following sections are included in the 2013 AEMP Report:

- Section 1 – Introduction;
- Section 2 – Site Characterization and Supporting Environmental Variables;
- Section 3 – Water Quality;
- Section 4 – Sediment Quality;
- Section 5 – Plankton;
- Section 6 – Benthic Invertebrate Community;
- Section 8 – Fish Community;
- Section 9 – Fish Tissue Chemistry;
- Section 10 – Fish Tasting;
- Section 11 – Special Studies (Littoral Special Study, Picoplankton Special Study, Downstream Lakes Special Study, Lake Trout Population Estimate Special Study, Stable Isotope Food Web Analysis Special Study);
- Section 12 – Weight of Evidence;
- Section 13 – Action Levels;
- Section 14 – Recommendations;
- Section 15 – Conclusions; and,
- Section 16 – Closure.

The fish health component of the AEMP is conducted every three years. It was conducted in 2012 and will be conducted again in 2015, and reported in the 2015 AEMP Annual Report. Section 7, Fish Health, is maintained as a placeholder for such reporting.

The Executive Summary provides a plain-language summary of the 2013 AEMP results for each of the above components.

1.6 Report Preparation

De Beers retained Golder Associated Ltd. (Golder) for the design, implementation, analyses, and reporting of the AEMP, with the exception of the fish tasting report (Section 10), which was prepared directly by De Beers. Golder has prepared this document in a manner consistent with the level of care and skill ordinarily exercised by members of the engineering and science professions currently practising under similar conditions in the jurisdiction in which the services are provided, subject to the time limits and physical constraints applicable to this document. No warranty, expressed or implied, is made.

This document, including all text, data, tables, plans, figures, drawings, and other documents contained herein, has been prepared by Golder for the sole benefit of De Beers. It represents Golder's professional judgement based on the knowledge and information available at the time of completion. Golder is not responsible for any unauthorized use or modification of this document. All third parties relying on this document do so at their own risk.

1.7 Report Limitations

The factual data, interpretations, suggestions, recommendation, and opinions expressed in this document pertain to the specific project, site conditions, design objective, development, and purpose described to Golder by De Beers for the Snap Lake Mine, and are not applicable to any other project or site location. This report is not intended to replace De Beers' standard operating procedures provided in the appropriate operation, maintenance, and surveillance manual or engineering design reports for each facility. To properly understand the factual data, interpretations, suggestions, recommendations, and opinions expressed in this document, reference must be made to the entire document.

1.8 References

De Beers (De Beers Canada Inc.). 2002. Snap Lake Diamond Project: Environmental Assessment Report. Submitted to the Mackenzie Valley Environmental Impact Review Board. Yellowknife, NWT, Canada.

De Beers. 2014. 2013 Aquatics Effects Monitoring Program Design Plan in Support of Water Licence (MV2011L2-0004). Snap Lake Project. Submitted to the Mackenzie Valley Land and Water Board. Yellowknife, NWT, Canada.

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MVLWB. 2004. Mackenzie Valley Land and Water Board Licence #MV2001L2-0002. Yellowknife, NWT, Canada.

MVLWB. 2013a. Mackenzie Valley Land and Water Board Updated Water Licence #MV2011L2-0004. Yellowknife, NWT, Canada.

MVLWB. 2013b. AEMP Design Plan and Re-evaluation Report – Plan Approval – De Beers Canada Inc. – Snap Lake Mine, Letter to Alexandra Hood (De Beers) from Willard Hagen (MVLWB), dated March 28, 2013.

MVLWB. 2013c. Aquatic Effects Monitoring Plan – Approval – De Beers' Aquatic Effects Monitoring Plan) AEMP Chapters 6 and 7 – De Beers – Snap Lake, Letter to Alexandra Hood (De Beers) from Willard Hagen (MVLWB), dated November 29, 2013.

WLWB (Wek´èezhìi Land and Water Board). 2010. Guidelines for Adaptive Management – A Response Framework for Aquatic Effects Monitoring - Draft. Yellowknife, NWT, Canada.

SECTION 2

SITE CHARACTERIZATION
AND ENVIRONMENTAL VARIABLES

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

LIST OF APPENDICES

LIST OF ACRONYMS

Term	Definition
ABS	absolute value
AEMP	Aquatic Effects Monitoring Program
De Beers	De Beers Canada Inc.
DSL1	Downstream Lake 1
DSL2	Downstream Lake 2
Golder	Golder Associates Ltd.
LCB	Lac Capot Blanc
LK13	Lake 13
Mine	Snap Lake Mine
n/a	not available
NAD	North American Datum
NEL	Northeast Lake
QA	quality assurance
QC	quality control
RPD	relative percent difference
SLMB	Snap Lake main basin
SLNW	Snap Lake northwest arm
SNAP	Snap Lake
UTM	Universal Transverse Mercator
WMP	water management pond
WTP	water treatment plant

UNITS OF MEASURE

Term	Definition
%	percent
°C	degrees Celsius
cm	centimetre
km/hr	kilometres per hour
L	litre
m	metre
m^3	cubic metre
m/s	metre per second
m^3/day	cubic metres per day
m^3/s	cubic metres per second
masl	metres above sea level
mm	millimetre
Mm^3	million cubic metres
W/m^2	Watts per square metre

2 SITE CHARACTERIZATION AND SUPPORTING ENVIRONMENTAL VARIABLES

2.1 Introduction

2.1.1 Background

The Site Characterization and Supporting Environmental Variables chapter summarizes information to describe the general conditions at the Snap Lake Mine (Mine) site and the local environment in which the Aquatic Effects Monitoring Program (AEMP) is conducted. The data presented in this chapter will assist in the interpretation of the component-specific AEMP results by the main AEMP components (i.e., water quality, sediment quality, plankton, benthic invertebrates, and fish community monitoring).

This chapter incorporates key information relevant to Snap Lake and its downstream and reference lakes aquatic environments. Information presented in this chapter includes Mine-related data (e.g., spills of process water or treated effluent, minewater discharge volumes, geochemistry data) and non-Mine related data (e.g., seasonal water temperature, hydrology, meteorological conditions [e.g., wind direction and speed, air temperature, and precipitation]).

Information on the environmental variables of the Mine site and its operations, as well as characteristics of the surrounding waterbodies were collected during monthly field programs (e.g., seasonal water temperature and ice thickness) or reported in the Environmental Assessment Report (De Beers 2002) and updated in annual reports and other documents prepared outside of the AEMP including:

- Monthly Surveillance Network Program reports;
- Air Quality, Meteorological Monitoring and Emissions Reporting 2013 Annual Report (Golder 2014a);
- Hydrology Annual Report (Golder 2014b);
- Acid Rock Drainage and Geochemical Characterization Report for the Water Licence (ARD Annual Report) (Golder 2014c);
- Water Licence Annual Report, and,
- Annual National Pollutant Release Inventory for the Mine.

2.1.2 Objective

The primary objective of the Site Characterization and Supporting Environmental Variables component is to provide a description of the Mine and non-Mine related modifying factors that may affect the Snap Lake ecosystem that need to be considered during data interpretation by each AEMP component.

Where available, the data collected at Snap Lake are compared to data from nearby reference lakes as agreed to in Water Licence MV2011L2-0004. The water temperature monitoring program compares data collected at Snap Lake to data from Northeast Lake and Lake 13 (Figure 2-1). The hydrology monitoring program compares data collected at Snap Lake to data from Northeast Lake, North Lake, and the 1999 Reference Lake (Figure 2-2). The key questions to be addressed by the Site Characterization and Supporting Environment Variables component are:

- What are the general conditions of the Mine site and the local environment in which the AEMP is conducted, independent of mining-related activities and considering unanticipated mining events such as spills?
- Is there a habitat difference between Snap Lake and the reference lakes in terms of seasonal water temperature and ice-cover?

2.2 Methods

2.2.1 General Site Condition Monitoring

The conditions at the Mine site in relation to the aquatic environment (e.g., spills of process water or treated effluent, minewater discharge) were characterized by reviewing information provided by De Beers Canada Inc. (De Beers) Mine site staff.

2.2.1.1 Spills

The reportable spill summary provided by De Beers was screened for reportable spills that occurred in or near a waterbody and of a volume considered to be large enough to possibly affect the aquatic environment.

2.2.1.2 Geochemistry

In accordance with Part E, Section 13 of the Water Licence #MV2011L2-0004, a seepage survey of the waste storage areas was conducted under the supervision of a Professional Geologist of the Northwest Territories specializing in acid rock drainage and hydrogeochemistry. Seeps were identified and communicated to De Beers for addition to the site water quality monitoring program. This information is reported on in the ARD Annual Report (Golder 2014c) submitted as a component of the Water Licence Annual Report. De Beers collected samples from the seeps according to the requirements of Part E, Schedule 4, Item 4:

> *"a) Sampling of detected seepages a minimum of twice per year (once during early summer freshet thaw and again in late summer or fall); additional monitoring should be conducted as soon as practicable following Major Storm Events;*
>
> *b) Each seepage survey shall include sampling at a reference location in an unaffected area;*

c) The monitoring plan shall include Action Levels for parameters of concern to trigger additional sampling or other activities;

d) Testing in the field shall include measurements of field pH, temperature, flow, conductivity, and observations of the physical properties of the seepage;

e) Laboratory analysis of each sample shall include major ions, Total Suspended Solids (TSS), Total Dissolved Solids (TDS), pH, total metals, and dissolved metals."

For the Site Characterization and Supporting Environmental Variables component of the AEMP, the seeps surveyed were screened for change from 2012 to 2013, having the potential to affect the aquatic environment of Snap Lake.

2.2.1.3 Project Description Changes

Site conditions encountered in 2013 with the potential to impact Snap Lake were summarized and compared by De Beers to the Consolidated Project Description for the Mine submitted to the Mackenzie Valley Land and Water Board in 2003 (De Beers 2003). The operational changes to the project description that may impact Snap Lake are not considered new information for regulators or stakeholders, but are summarized for consideration by AEMP components.

2.2.1.4 Volume of Treated Effluent Discharged

Minewater, including water used for camp and underground operations, as well as seepage and runoff water from the North Pile and the rest of site, is captured within the water control structures and pumped into the water treatment plant (WTP); domestic waste water is pumped into the sewage treatment plant and then the WTP. Treated effluent from the WTP is discharged daily into Snap Lake through three minewater outlet pipelines equipped with diffusers (herein referred as diffuser pipes). One permanent diffuser pipe discharged treated effluent from the WTP throughout 2013. To prevent treated effluent spills by allowing additional discharge, a temporary diffuser pipe was used from May 18, 2013 to October 5, 2013. A second permanent diffuser was installed in the fall 2013 and started discharging treated effluent October 6, 2013.

Daily volumes of treated effluent from the WTP discharged into Snap Lake were provided by De Beers site staff[1]. These daily discharge volumes were used to determine monthly trends in 2013.

2.2.2 Meteorological Monitoring

During 2013 meteorological data, including rainfall, temperature, wind, relative humidity, and solar radiation, were collected at the hill meteorological monitoring station (Hill Station), located on an elevated

[1] Information provided by De Beers, January 20, 2013

point of land immediately west of the WTP, and at the lake hydro-meteorological monitoring station (Lake Station) located northeast of the Mine, close to Snap Lake (Figure 2-1). Meteorological data will be reported in the Annual Air Quality, Meteorological Monitoring and Emissions Report (Golder 2014a), and are summarized herein.

The meteorological data were collected, reviewed and interpreted by the authors of the Annual Air Quality, Meteorological Monitoring and Emissions Report (Golder 2014a). The data collected at Snap Lake were compared to the Environment Canada data collected in Yellowknife (Environment Canada 2013).

Likely due to equipment malfunction, the Lake Station appeared to overestimate total precipitation (snow and rainfall) in comparison to average precipitation of the area. As a result, the Lake Station precipitation data were considered not valid and were screened out of the data set.

2.2.3 Hydrological Monitoring

2.2.3.1 Lake Elevations and Survey Benchmarks

At each stream monitoring station, water elevation was measured relative to an established benchmark to allow for continuity between yearly data sets (Golder 2014b). Benchmarks were established by setting metal pins into bedrock and surveying the pins for elevation in metres above sea level (masl). Benchmarks were installed at Snap Lake (H3), Snap Lake Outflow (H1 and H2), Snap Lake inflow (H4), North Lake, Northeast Lake, and 1999 Reference Lake (Figure 2-2 and Table 2-1).

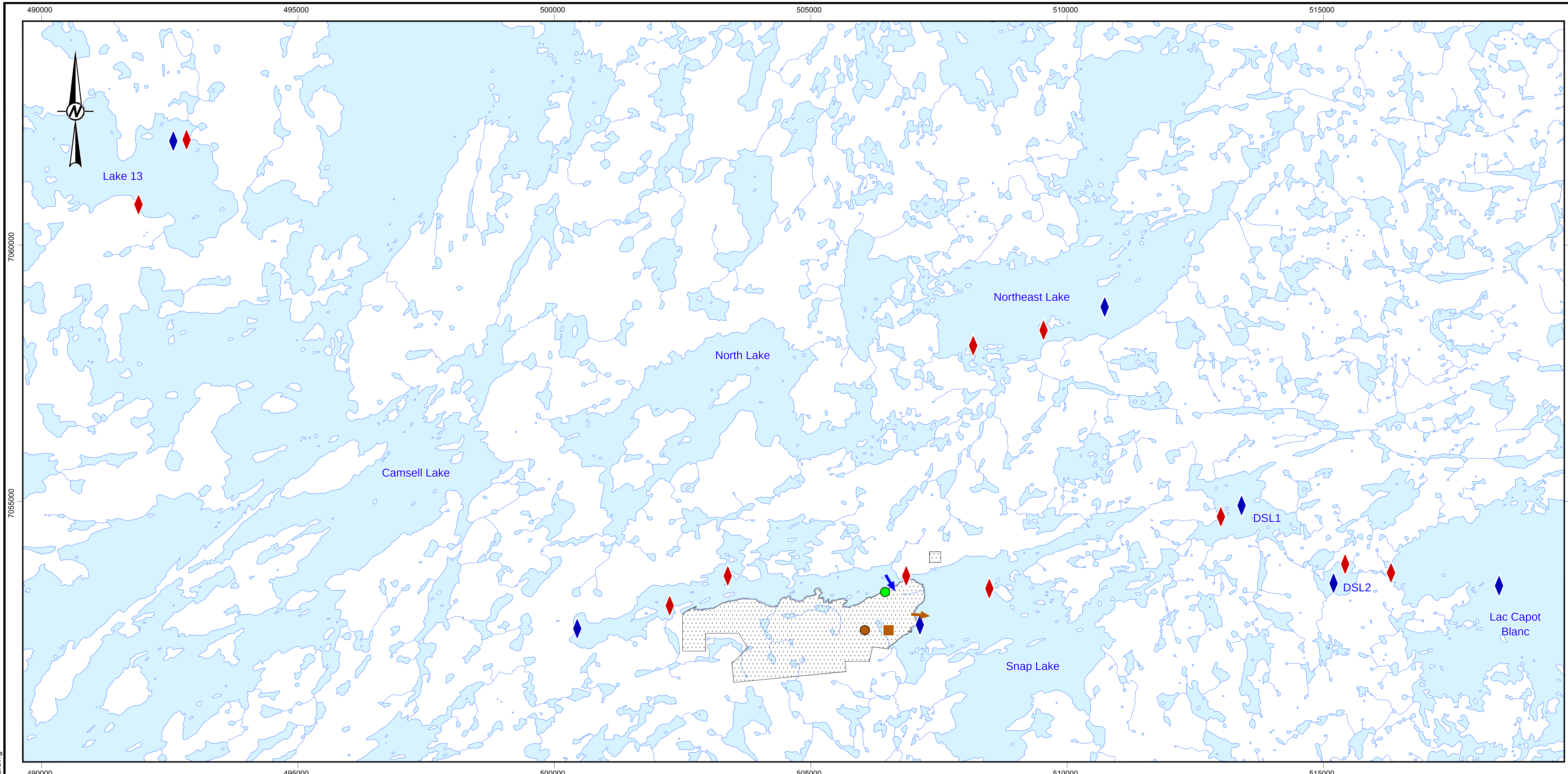
When stream discharge was assessed at each monitoring station, the water elevation was measured relative to the established benchmark using an engineer’s rod and level. The relationship between the water elevation (stage) in the channel and flow (discharge) was established using the Aquarius hydrologic software package, which produces a rating table showing calculated discharges for a defined range of stages. Using this rating table, stage data points collected using water level data loggers (Solinst Levelogger Gold 3001) were converted to point discharge values, allowing discharge volumes to be calculated per hour, day, and month.

Table 2-1 Benchmark Locations and Elevations

Station Designation	UTM (NAD 83, Zone 12)		Geodetic Elevation (masl)
	North	East	
H1 – Snap Lake Outflow	7054115	512105	444.341
H2 – Snap Lake Outflow	7053946	512231	443.842
H3 – Snap Lake Elevation[(a)]	7051483	506811	444.840
H4 – Snap Lake Inflow[(a)]	7051483	506811	444.840
North Lake	7056652	507682	440.720
Northeast Lake	7063614	510192	433.641
1999 Reference Lake	7042237	496879	441.492

a) H3 and H4 surveyed from same benchmark location.

UTM = Universal Transverse Mercator; NAD = North American Datum; masl = metres above sea level.



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LEGEND

- Watercourse
- Snap Lake Mine Footprint
- Waterbody
- Meteorological Lake Sampling Station
- Meteorological Hill Sampling Station
- Temperature Logger Location - Shallow
- Temperature Logger Location - Deep
- Diffuser Pipes
- Snap Lake Water Intake
- Water Treatment Plant

REFERENCES

Base Data Source: CanVec

NOTES

DSL1 = First Lake Downstream of Snap Lake

DSL2 = Second Lake Downstream of Snap Lake

SNAP LAKE MINE

Temperature Logger and Meteorological Stations Locations, 2013

PROJECTION: UTM Zone 12	DATUM: NAD83	DE BEERS GROUP OF COMPANIES	
1.5 0 1.5 SCALE 1:75,000 KILOMETRES			
FILE No: 1313490001-7010_2-1		DATE: March 11, 2014	
JOB No: 14-1349-0004	REVISION No: A	Figure **2-1**	
OFFICE: GOLDER VICTORIA	DRAWN: JEF	CHECK: PMC	

LEGEND

Watercourse

Snap Lake Mine Footprint

Waterbody

Hydrological Monitoring Station

NOTES

DSL1 = First Lake Downstream of Snap Lake

DSL2 = Second Lake Downstream of Snap Lake

REFERENCES

Base Data Source: CanVec

SNAP LAKE MINE

Hydrological Monitoring Stations, 2013

PROJECTION:	DATUM:
UTM Zone 12	NAD83

2.5 0 2.5

SCALE 1:125,000 KILOMETRES

DE BEERS
GROUP OF COMPANIES

FILE No:	DATE:
1313490001-7010_2-2	February 24, 2014

JOB No:	REVISION No:		
14-1349-0004	A		
OFFICE:	DRAWN:	CHECK:	
GOLDER VICTORIA	JEF	PMC	

Figure **2-2**

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2.2.3.2 Stream Discharge

Stream velocities for all monitoring locations were measured using a Swoffer Model 2100 current meter or a Marsh McBirney Flo-Mate model 2100 flow meter attached to a top setting wading rod (Golder 2014b). A tag line marked at 0.1 metre (m) intervals was used to measure the width of all channels. The tag line was attached to sections of rebar driven into the stream banks. The channels were divided into vertical segments of approximately 5 percent (%) of the channel width. For the H1 and H2 flumes, depth and velocity profiles were taken across the flume width. Velocity and depth were measured at the centre of each segment. For water depths less than or equal to 0.7 m, the velocity was measured at a depth of 60% of the total depth from the surface. For water depths greater than 0.7 m, standard operating procedure dictates that the velocity be measured at 80% and 20% of depth, and the measured velocities averaged; however, all discharge point water depths were below 0.7 m in 2013.

For each segment, discharge (in cubic metres per second [m^3/s]) was calculated as follows:

Segment discharge (m^3/s) = Depth (m) x Velocity (m/s) x Width (m)

Where: **m^3/s** = cubic metres per second;
m = metre;
m/s = metres per second;
Depth = depth measured at the centre of each segment;
Velocity = mean velocity measured at the centre of each segment; and,
Width = distance measured between the centre points of adjacent segments.

Total discharge of the stream was then calculated by summing the measured segment discharges as per the mid-section method (Raghunath 2006).

2.2.3.3 Continuous Water Level Recording

Water surface elevations were recorded every 30 minutes using Solinst Levelogger Gold 3001 data loggers from June 2, 2013 to September 16, 2013 at Snap Lake outflow stations H1 and H2, and from June 2, 2013 to September 17, 2014 at Snap Lake water surface elevation station H3. The water surface elevation for Snap Lake was also surveyed at the 2005 Benchmark by Nampcy Solutions Ltd. using a rod and level daily between May 30 and August 21, 2013, and 21 times between August 27 and October 14, 2013. The water surface elevation at H4 was recorded every 30 minutes from May 27, 2013 to September 17, 2013.

At the Snap Lake outflow stations (H1 and H2), the Leveloggers are mounted to brackets installed in the streambed. At the Snap Lake station (H3), the Levelogger is located in water approximately 1.2 m deep and about 5 m from shore. At the Snap Lake inflow station (H4), the logger is located mid-stream in less than 1 m of water.

2.2.4 Seasonal Water Temperature

To determine lake temperatures during the open water season, temperature loggers (TidbiT Water Temperature Data Loggers - UTBI-001), also referred to as thermographs, were installed in early July 2013 (Table 2-2) in two areas on Snap Lake and the following study lakes (Figure 2-1):

- Snap Lake main basin (SLMB);
- Snap Lake northwest arm (SLNW);
- Northeast Lake;
- Lake 13;
- Downstream Lake 1 (DSL1);
- Downstream Lake 2 (DSL2); and,
- Lac Capot Blanc (LCB).

Thermographs were installed in shallow habitat sites (i.e., total depth of approximately 1.0 m) and in deep basins (i.e., depths ranging from 13.6 to 39.8 m) at each of the study lakes (Table 2-3; Figure 2-1). Water temperature was recorded hourly until retrieval of the thermographs in September 2013.

Table 2-2 Snap Lake Mine Shallow Habitat and Deep Basin Temperature Logger Installation and Retrieval Dates, 2013

Study Lake	Installation Date	Retrieval Date	Comments
SLMB	July 5, 2013	September 5, 2013	
SLNW	July 5 and 6, 2013	September 8, 2013	On July 5 one shallow habitat and the deep basin temperature loggers were installed. The second shallow habitat temperature logger was installed on July 6.
NEL	July 11, 2013	September 14 and 15, 2013	Deep basin temperature loggers were retrieved on September 14. Shallow habitat temperature loggers were retrieved on September 15.
LK13	July 6, 2013	September 14 and 15, 2013	One shallow habitat and the deep basin temperature loggers were retrieved on September 14. One shallow habitat temperature logger was retrieved on September 15.
DSL1	July 11, 2013	September 7, 2013	
DSL2	July 11, 2013	September 10, 2013	
LCB	July 13, 2013	September 10, 2013	

SLMB = Snap Lake main basin; SLNW = Snap Lake northwest arm; NEL = Northeast Lake; LK13 = Lake 13; DSL1 = Downstream Lake 1; DSL2 = Downstream Lake 2; LCB = Lac Capot Blanc.

2.2.4.1 Shallow Habitat Temperature

A total of 11 temperature loggers were installed in shallow habitat sites: two each in SLMB, SLNW, Northeast Lake, and Lake 13, and one each in DSL1, DSL2, and LCB (Figure 2-1). The thermographs were installed approximately 0.5 m below the water surface on a mooring (i.e., steel cable held by an anchored buoy). For SLMB, SLNW, Northeast Lake, and Lake 13, the average temperature of both shallow habitat sites was reported.

2.2.4.2 Deep Basins Temperature

A series of temperature loggers suspended on a mooring was installed in one deep basin of SLMB, SLNW, Northeast Lake, Lake 13, DSL1, DSL2, and LCB. The deep stations coincide with AEMP stations (Table 2-3, Figure 2-1). In DSL1, DSL2, and LCB, two thermographs were installed per mooring, targeted for 0.5 m above substrate and at 0.3 m below water surface (Table 2-3).

To support the Lake Trout Population Estimate Special Study (Chapter 11, Section 3), a series of thermographs was installed at 3 m depth intervals in SLMB, SLNW, Northeast Lake, and Lake 13 (Table 2-3, Figure 2-1) to quantify the volume of useable habitat by Lake Trout (i.e., water temperatures below 15 degrees Celsius [°C]) in each lake.

Table 2-3 Snap Lake Mine Deep Basin Temperature Logger Locations, 2013

Deep Basin Location	Associated AEMP Station	Basin Depth (m)	# Temperature Loggers/Mooring	Comments
SLMB	SNAP02-20E	31.8	11	
SLNW	SNAP20B	39.8	14	
NEL	NEL-06	27.0	9	Surface logger installed at 2.5 m from surface because field conditions did not allow the field crew to adjust the logger on the mooring. Logger installed at 15.5 m depth was non-functional when retrieved and no data were recovered.
LK13	LK13-03	18.8	8	
DSL1	DSL1-1	13.6	2	Surface logger installed at 0.7 m; bottom logger at 0.5 m from substrate.
DSL2	DSL2-2	7.0	2	Surface logger installed at 0.4 m; bottom logger at 0.5 m from substrate.
LCB	LCB-1	14.0	2	Surface logger installed at 1.1 m; bottom logger at 0.5 m from substrate.

Note: Because substrate is composed mostly of organic material, it is possible that the anchors sank into the substrate, shifting the temperature loggers' depth down closer to the substrate.

SLMB = Snap Lake main basin; SLNW = Snap Lake northwest arm; NEL = Northeast Lake; LK13 = Lake 13; DSL1 = Downstream Lake 1; DSL2 = Downstream Lake 2; LCB = Lac Capot Blanc; SNAP = Snap Lake; AEMP = Aquatic Effects Monitoring Program; m = metre; # = number.

2.2.5 Ice Thickness and Ice Cover Monitoring

Ice thickness measurements were collected monthly at Snap Lake from 2005 to 2013, at Northeast Lake from 2008 to 2013, and at Lake 13 in May 2013. Average yearly and monthly ice thickness was calculated for each lake. Ice thickness measurements are not available for LCB, DSL1, and DSL2 because it was not required under Water Licence MV2011L2-0004.

Days of ice cover were determined for Snap Lake from De Beers site staff observations. From 2008 to 2012, the first day of ice cover (Ice-On date) was considered to be the date in which a layer of ice was observed on the main basin of Snap Lake, and the last day of ice cover (Ice-Off date) was considered to be the day in which the main basin ice layer melted. In 2013, Ice-On and Ice-Off dates were observed on the west arm of Snap Lake and not the main basin.

No information on days of ice cover is available for the remote reference lakes and the downstream lakes, as daily observations from site staff would be required to obtain this information, which is not logistically practicable.

2.3 Quality Assurance and Quality Control

The objective of the Quality Assurance (QA)/ Quality Control (QC) program is to standardize methods so that field sampling, data entry, data analyses, and report preparation produce technically sound and scientifically defensible results.

Meteorological data, hydrology data, geochemistry data, and Mine operations information (e.g., spills, treated effluent discharge) presented in this section have undergone QA/QC review as part of their respective disciplines.

Water temperatures collected with the temperature loggers were compared to field measurements collected during installation and retrieval of the loggers, and during the monthly AEMP field programs when available, to check for accuracy and consistency. The temperature data collected on the day of installation and on the day of retrieval of the water temperature loggers were removed from the data set to allow uniformity of the data presented and to prevent the use of data affected by handling of the loggers by the field crew. Standard deviations were used to measure temperature variability in one day, to detect drastic temperature changes. When two temperature loggers were installed at shallow locations (i.e., SLMB, SLNW, Northeast Lake, and Lake 13), the relative percent difference (RPD) was calculated to compare readings of both loggers and check that the loggers measured approximately the same temperatures. The RPD was calculated using the following formula:

$$RPD = ABS\ \frac{(\text{temperature of logger 1} - \text{temperature of logger 2})}{\text{average temperature of loggers 1 and 2}} \times 100$$

where ABS = absolute value.

Temperature variability between the two readings was assessed as notable if it was greater than 20% and for more than 10% of the readings in a day. The average of both readings was reported.

Ice thickness data have been collected since 2005 for Snap Lake, since 2008 for Northeast Lake, and once in May 2013 for Lake 13. The QA/QC of ice thickness data involved several stages of “spot checks” to maintain accuracy and consistency.

2.4 Results

2.4.1 General Site Condition Monitoring

2.4.1.1 Spills

The De Beers site staff recorded 10 reportable spill events in 2013. The complete list was reviewed to identify reportable spills at the Mine site that occurred close (i.e., less than 50 m) to Snap Lake or any other waterbody. One such spill event was identified (Table 2-4) and spill mitigation was applied. This reported spill did not enter Snap Lake, and is not anticipated to have a measurable effect on the aquatic environment.

Table 2-4 Spill event at the Snap Lake Mine, 2013

Date	Product	Volume (L)	Location	Description
May 17, 2013	Process water	200	South of Perimeter Sump 3	An environmental sampling port on the underside of the other 6" pump line was open and not initially detected; leakage occurred with the backflow.

Source: “2014 Information Request” spreadsheet provided by De Beers staff, January 20, 2013.

L = litre.

2.4.1.2 Geochemistry

No substantial changes were observed during the 2013 seepage surveys of the waste storage areas compared to 2012 (Golder 2014c). The results of the 2013 water quality analyses at most seepage monitoring stations were similar to concentration trends observed in 2012. Seepage at the Mine in 2013 is not anticipated to have had a measurable effect on the aquatic environment.

2.4.1.3 Project Description Changes

The Consolidated Project Description for the Mine was submitted to the Mackenzie Valley Land and Water Board in 2003 (De Beers 2003). Table 2-5 summarizes the February 2013 and current (i.e., up to end of 2013) site conditions that differ from the original project description and that might impact Snap Lake.

Table 2-5 Project Description Changes, 2013 and Current Conditions

Topic	Original Project Description	2013 Site Conditions[(a)]	Current Site Conditions[(b)]
Freshwater	Freshwater will be drawn from Snap Lake and used in the Process Plant and as drill water for underground drilling.	No freshwater, but only treated effluent from the WTP was used in the Process Plant in 2013. Freshwater was drawn from Snap Lake only for domestic use, fire suppression, and exploration drill water (surface).	Minimal freshwater is now used in the Process Plant and freshwater is also used for domestic use, fire suppression, and exploration drill water (surface).
Site runoff	Rockfill ditches and grading will direct the runoff from the peninsula areas towards the WTP.	Rockfill ditches and grading direct runoff from the peninsula areas to the WMP, not the WTP.	Rockfill ditches and grading direct runoff from the peninsula areas to the WMP, not the WTP.
North Pile, landfill and landfarm runoff external water collection system	All water entering the ditches surrounding the perimeter of the North Pile will be pumped to the WTP. Runoff from the landfill and landfarm will be pumped to the WTP.	Runoff water, except for the South Pit water, is pumped to the WMP, not directly to the WTP. Ditches are now referred to as sumps. Runoff from the landfill and land farm is pumped to the WMP and then to the WTP	Runoff water, except for the South Pit water, is pumped to the WMP, not directly to the WTP. Runoff from the landfill and land farm is pumped to the WMP and then to the WTP.
Dust suppression	Water for dust suppression of the North Pile will be drawn from the WTP at a rate of 55 m^3/day for six months per year.	The North Pile was not sprayed for dust suppression, as it was not necessary.	The North Pile is not sprayed for dust suppression, as it was not necessary.
Dam raises	Dam 1 and Dam 2 will be raised by 2 m each to increase capacity.	There were no dam raises to date.	There have been no dam raises to date.
Sediment loads in runoff water	Runoff from the North Pile and core site facilities will be sent directly to the filter feed tank since suspended solids will be low.	Runoff from the North Pile and core site facilities was sent to the WMP.	A sedimentation / floc tank has been added upstream of the WMP to pre-treat runoff from the North Pile.
Domestic waste water treatment	The sludge from the sewage treatment plant will be incinerated and placed in the landfill.	The sludge from the sewage treatment plant was placed in the landfill or incinerated and then placed in the landfill.	The sludge from the sewage treatment plant is still deposited in the landfill or incinerated and then placed in the landfill.

a) Project Description Changes provided by De Beers, February 11, 2013.

b) Project Description Changes provided by De Beers, and current up until end of 2013 calendar year. WMP = water management pond; TWP. = water treatment plant; m = metre; m^3/day = cubic metres per day.

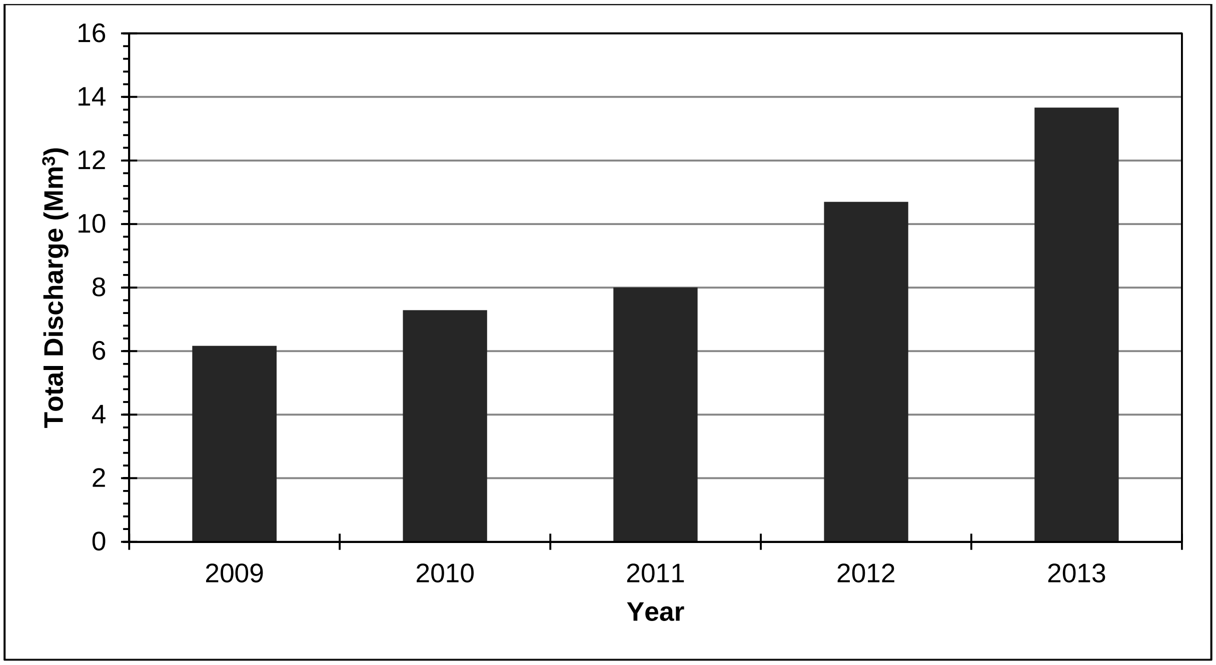
2.4.1.4 Volume of Treated Effluent Discharged

Total annual treated effluent discharge volumes from 2009 to 2013 are presented in Figure 2-3. The 2013 daily treated effluent discharge volumes are shown in Figure 2-4. Monthly volumes of treated effluent discharged in 2013[2] from the WTP into Snap Lake are summarized in Table 2-6. A total of 13.7 million

[2] Information provided by De Beers, January 20, 2014.

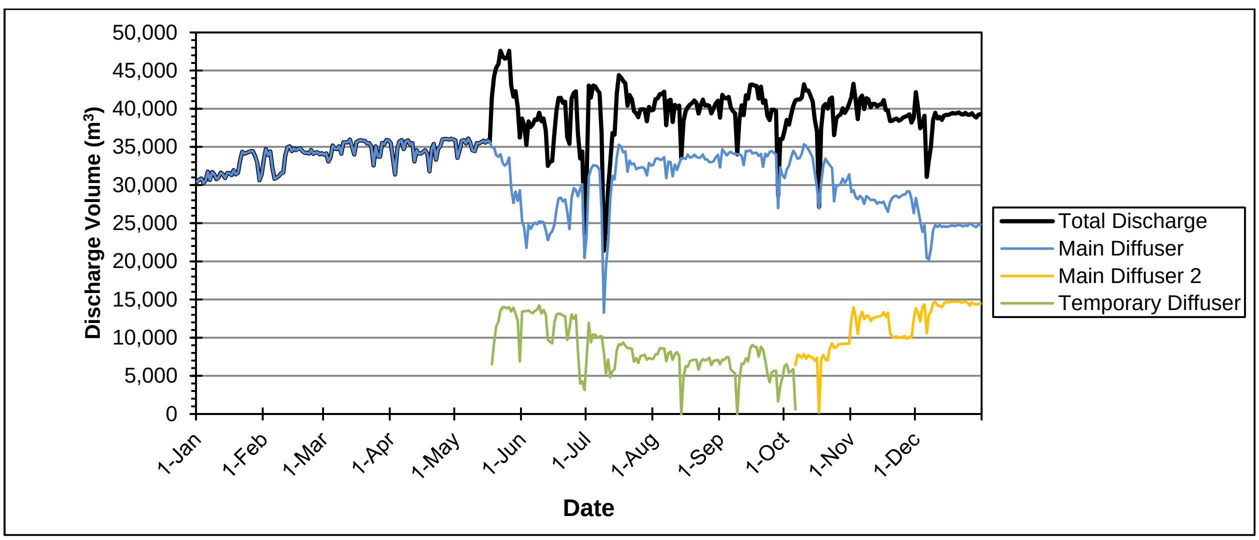
cubic metres (Mm^3) of treated effluent were discharged in 2013, which is an increase of 28% over the 10.7 Mm^3 discharged in 2012.

Figure 2-3 Annual Treated Effluent Discharge into Snap Lake, 2009 to 2013



Mm^3 = million cubic metres.

Figure 2-4 Daily Treated Effluent Discharge into Snap Lake, 2013



m^3 = cubic metre; Jan = January; Feb = February, Mar = March; Apr = April; Jun = June; Jul = July; Aug = August; Sep = September; Nov = November; Dec = December.

Modifications to the existing treated effluent discharge system increased the daily treated effluent discharge volume by approximately 8,000 cubic metres per day (m^3/day) in 2013 compared to 2012. The temporary floating diffuser provided an average additional discharge capacity of 8,460 m^3/day between May 18 and October 6, 2013. The second main diffuser provided an average additional discharge capacity of 11,406 m^3/day between October 6 and December 31, 2013. The minimum and maximum discharges occurred respectively in July and in the last week of May 2013 (Table 2-6; Figure 2-4).

Table 2-6 Treated Effluent Discharge Volumes from the Snap Lake Mine, 2013

Month of Discharge (2013)	Average Discharge (m^3/day)	Minimum Discharge (m^3/day)	Maximum Discharge (m^3/day)	Total Discharge (m^3)
January	31,972	30,242	34,438	**991,140**
February	33,732	30,806	35,079	**944,486**
March	34,935	32,565	35,954	**1,082,994**
April	34,792	31,376	36,070	**1,043,766**
May	39,249	33,552	47,596	**1,216,715**
June	37,396	23,627	42,293	**1,121,868**
July	38,753	21,376	44,399	**1,201,354**
August	40,199	33,546	42,239	**1,246,156**
September	39,853	28,613	43,191	**1,195,596**
October	39,559	27,113	43,209	**1,226,321**
November	39,930	38,165	43,276	**1,197,907**
December	38,610	31,061	42,179	**1,196,918**
Total 2013	**37,439**	**21,376**	**47,596**	**13,665,221**

m^3/day = cubic metres per day, m^3 = cubic metre.

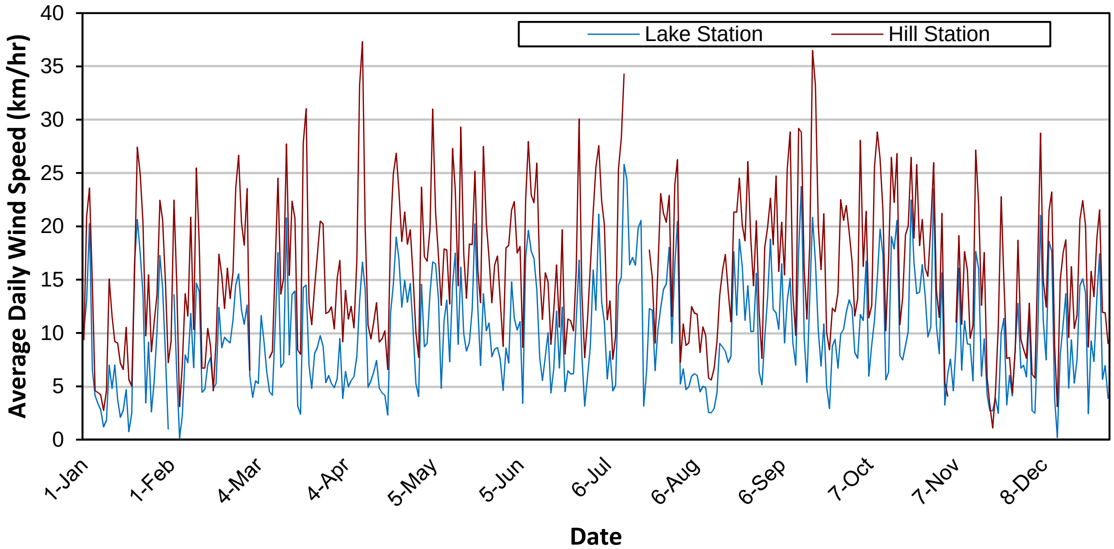
2.4.2 Meteorological Monitoring

Wind conditions, relative humidity, air temperature, and solar radiation for the Hill and Lake stations, as well as rainfall for the Hill Station in 2013 are presented in Figures 2-5 to 2-11 and Appendix 2A, Figures 2A-1 to 2A-6

2.4.2.1 Wind Speed and Direction

Hill Station is located on top of a hill and is more open to the wind; average daily wind speed was higher at Hill Station than at Lake Station (Figure 2-5). Both stations followed the same wind speed trend throughout the year. Daily average wind speed varied greatly; during open water season (June to October) days of peak wind speed, here defined as days when wind speed was both above 25 kilometres per hour (km/hr) for the Hill Station and above 20 km/hr for the Lake Station, were observed on: July, 3, 12, and 31; September 13 and 17; and, October 17, 22, and 30, 2013 (Figure 2-5).

Figure 2-5 Daily Lake and Hill Station Wind Speed, 2013



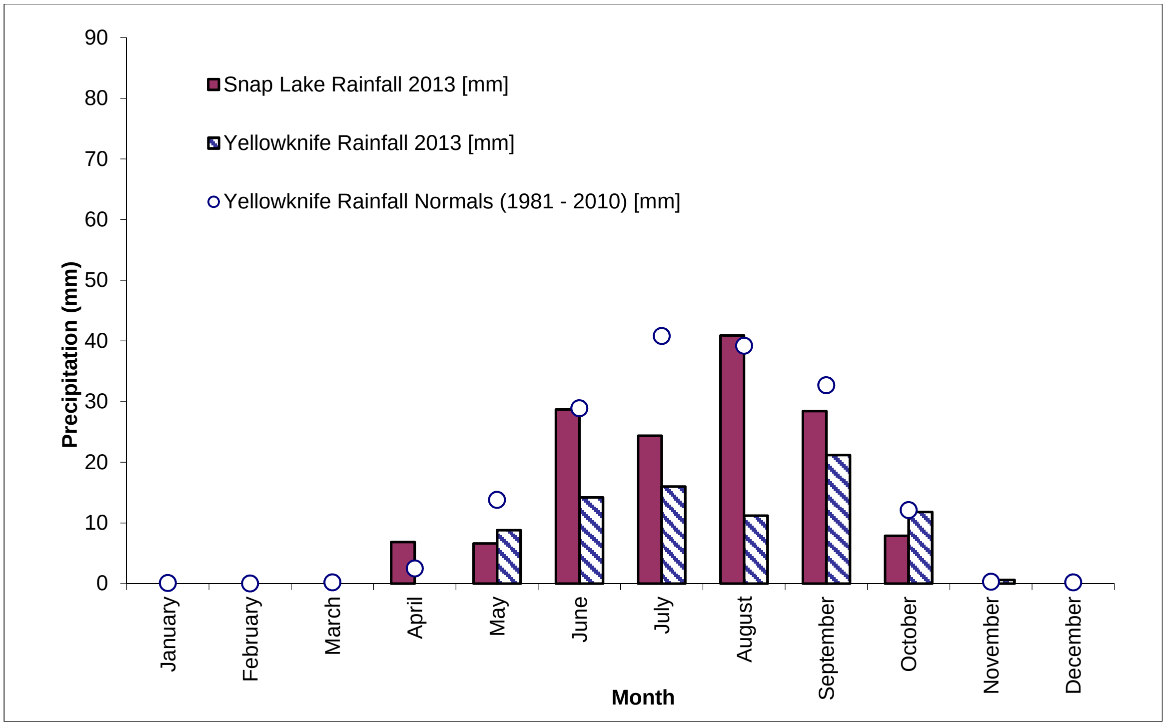
km/hr = kilometres per hour; Jan = January; Feb = February, Mar = March; Apr = April; Jun = June; Jul = July; Aug = August; Sep = September; Nov = November; Dec = December.

Similar to previous years, predominant winds at the Mine in 2013 were from the east and east-southeast (Appendix 2A, Figures 2A-1 to 2A-4). Lower wind speeds were measured from the south-west. In total, the Lake Station measured more calms (wind speed below 10 km/hr) than the Hill Station.

2.4.2.2 Rainfall

The 2013 rainfall monthly trends for the Hill Station are presented in Figure 2-6. The total annual rainfall recorded at the Hill Station for Snap Lake in 2013 was 143.8 millimetres (mm), which is approximately 3.6% higher than in 2012 (138.7 mm), 73.6% higher than the Yellowknife total for 2013 (83.8 mm), and 15.8% lower than the Yellowknife long-term (1981 to 2010) annual rainfall average of 170.8 mm (Environment Canada 2013). Monthly rainfall generally followed the same pattern observed in Yellowknife from 1981 to 2010 (Figure 2-6).

Figure 2-6 Hill Station Rainfall, 2013



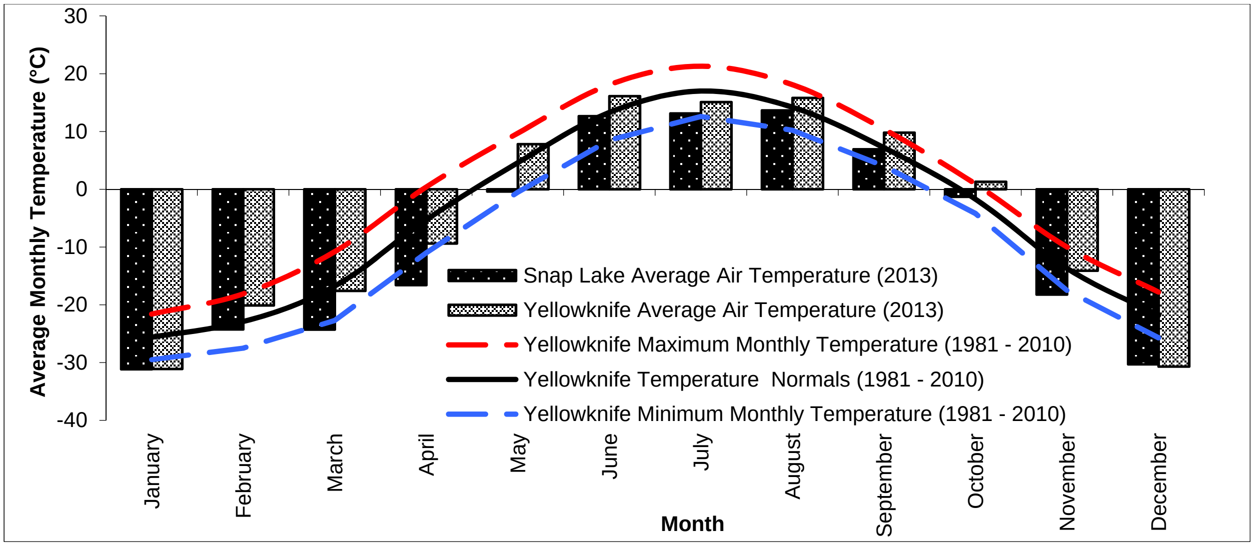
mm = millimetre.

2.4.2.3 Air Temperature

The 2013 average annual air temperature was -8.21°C at the Hill Station and -8.35°C at the Lake Station. These temperatures were approximately 1.5°C colder than measured in 2012, and the average annual temperature measured at Snap Lake (including Hill and Lake stations) from 2003 to 2012. These temperatures were in average 4.0°C colder than the annual temperature of -4.3°C for Yellowknife during 1981 to 2010.

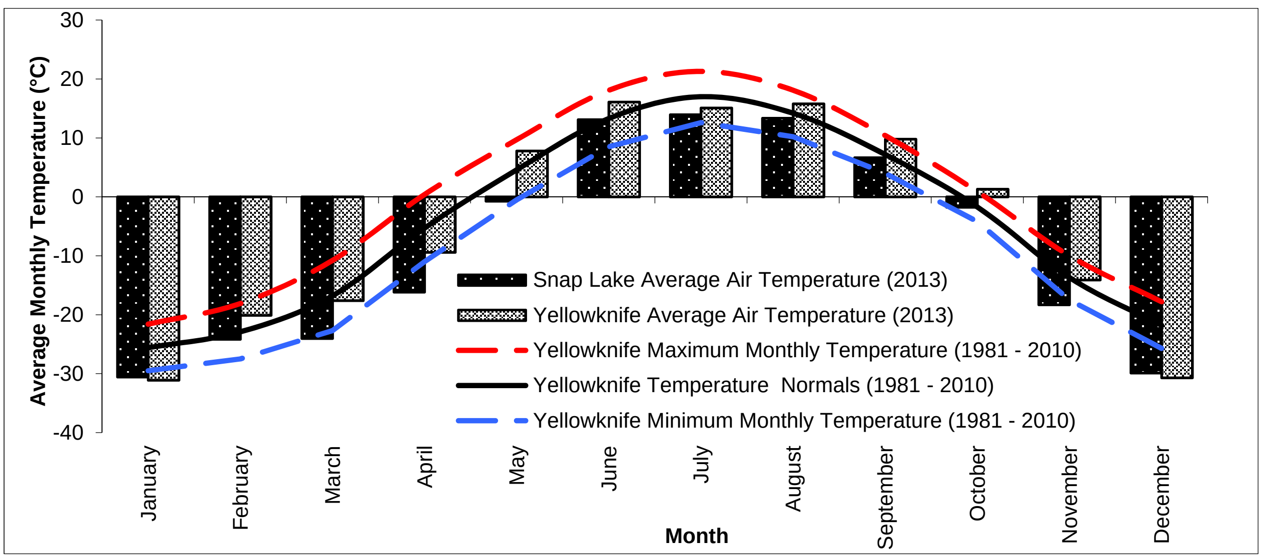
Yellowknife was 1.6°C colder in 2013 than in 2012, and 0.5°C colder in 2013 than the long-term average of -4.3°C (1981 to 2010). Monthly temperature data for the Lake and Hill stations are presented in Figures 2-7 and 2-8.

Figure 2-7 Lake Station Air Temperature, 2013



°C = degrees Celsius.

Figure 2-8 Hill Station Air Temperature, 2013



°C = degrees Celsius.

2.4.2.4 Solar Radiation

Solar radiation measured at the Lake and Hill stations followed the same trend as air temperature (Table 2-7 and Figure 2-9). Hill Station measures total solar radiation (i.e., all radiation coming in) while Lake Station measures the net solar radiation (i.e., all radiation coming in, minus all radiation reflected by ground/water, thermal radiation loss). The total and net radiations are two different parameters and are not comparable. As expected, Lake Station net solar radiation was negative from January to April and from October to December 2013, because the sum of the reflected radiation and the thermal radiation was higher than the radiation coming in (Table 2-7, Figure 2-9).

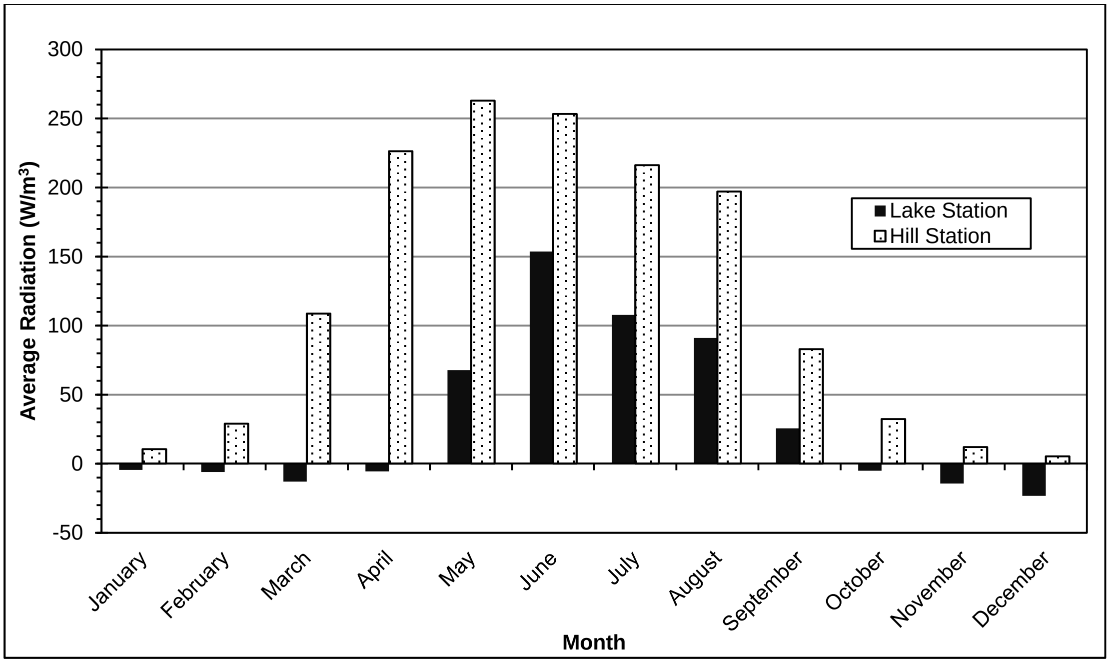
Table 2-7 Solar Radiation at the Lake and Hill Stations, 2013

Month	Lake Station			Hill Station		
	Radiation Hours	Average Radiation (W/m²)	Net Solar Radiation (W/m²)	Radiation Hours	Average Radiation (W/m²)	Total Solar Radiation (W/m²)
January	724	-4.49	**-3,247**	744	10.60	**7,889**
February	651	-5.98	**-3,892**	672	28.98	**19,474**
March	739	-12.99	**-9,601**	744	108.66	**80,841**
April	720	-5.47	**-3,940**	720	226.29	**162,932**
May	744	67.86	**50,489**	744	262.86	**195,568**
June	720	153.69	**110,657**	720	253.25	**182,340**
July	744	107.75	**80,166**	742	216.27	**160,471**
August	744	91.04	**67,734**	714	197.12	**140,745**
September	720	25.58	**18,420**	697	83.02	**57,865**
October	744	-5.09	**-3,787**	743	32.32	**24,017**
November	717	-14.28	**-10,238**	720	12.10	**8,711**
December	744	-23.30	**-17,335**	744	5.36	**3,986**
2013	**8,711**	**31.19**	**275,426**	**8,704**	**119.74**	**1,044,840**

Notes: net solar radiation does not include reflected radiation (i.e., reflection from ground, water, thermal radiation loss). Total solar radiation includes all radiation.

W/m² = Watts per square metre.

Figure 2-9 Average Solar Radiation at the Lake and Hill Stations, 2013

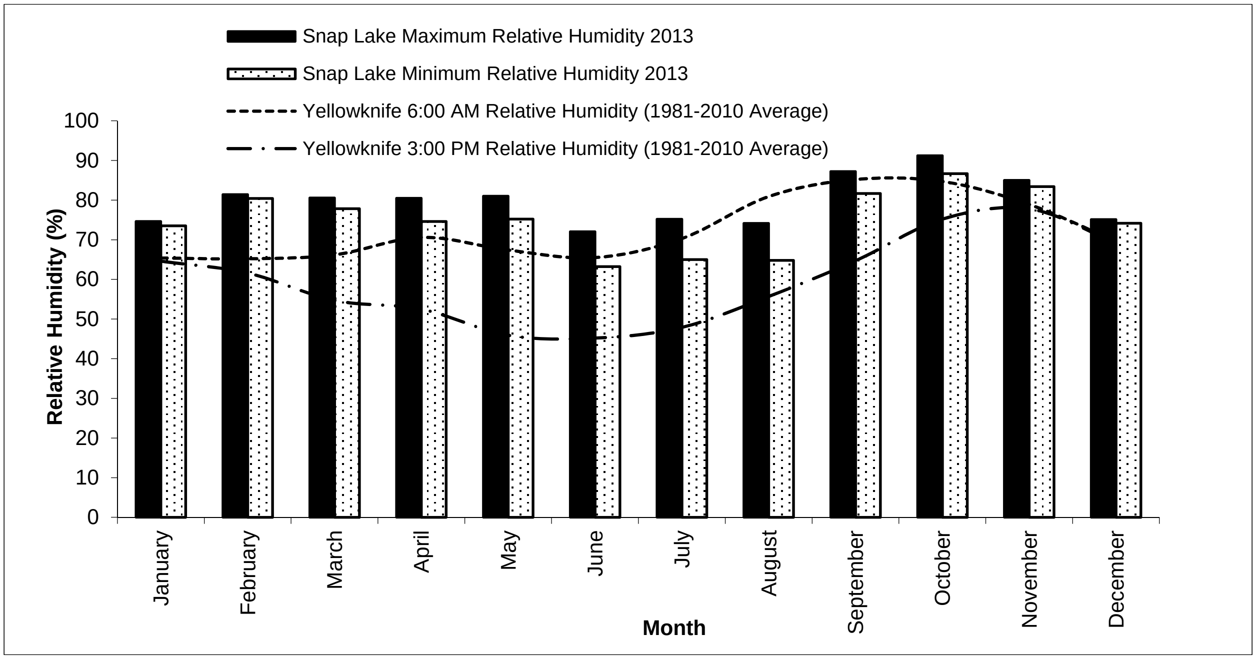


W/m² = Watts per square metre.

2.4.2.5 Humidity

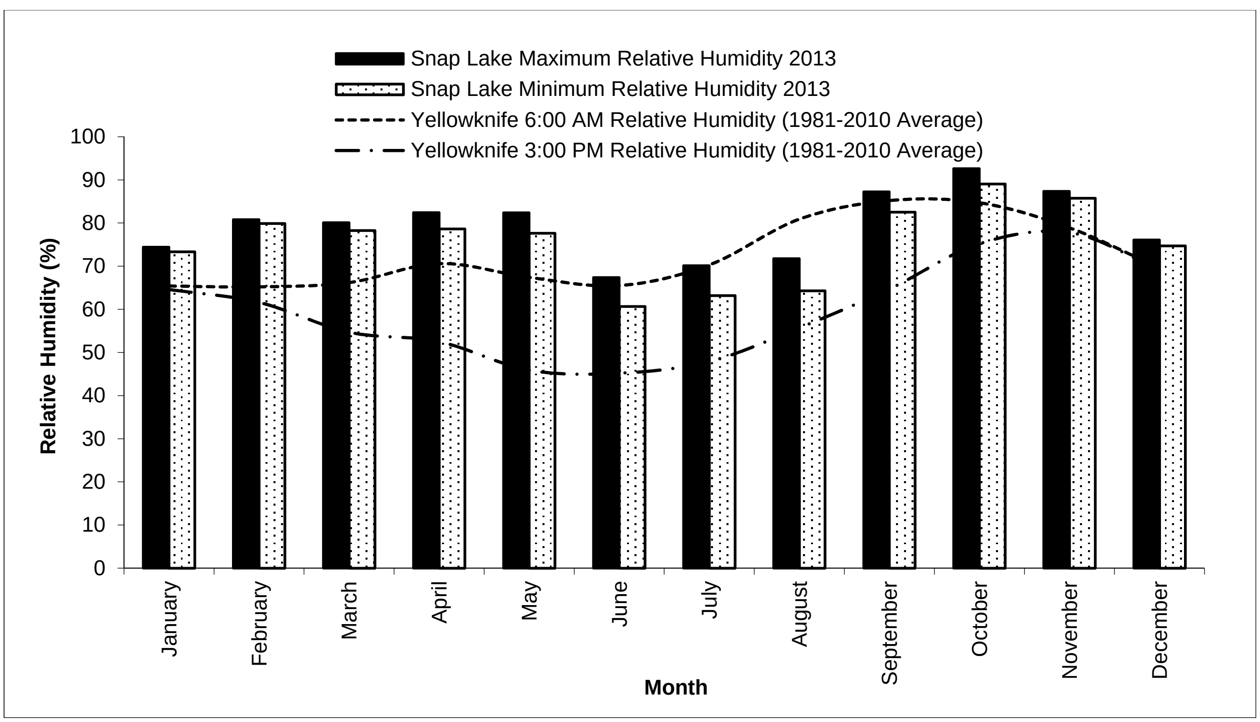
The data for relative humidity for the Snap Lake Hill and Lake Stations are consistent with the patterns and ranges of the Yellowknife averages from 1981 to 2010 (Figures 2-10 and 2-11). The relative humidity data are higher on average at Snap Lake than Yellowknife, which could be attributed to overall slightly lower ambient temperatures, but similar levels of absolute ambient moisture. The average relative humidity data for the Snap Lake Hill and Lake Stations are presented in Appendix 2A, Figures 2A-5 and 2-A6.

Figure 2-10 Lake Station Relative Humidity, 2013



% = percent.

Figure 2-11 Hill Station Relative Humidity, 2013



% = percent.

2.4.3 Hydrological Monitoring

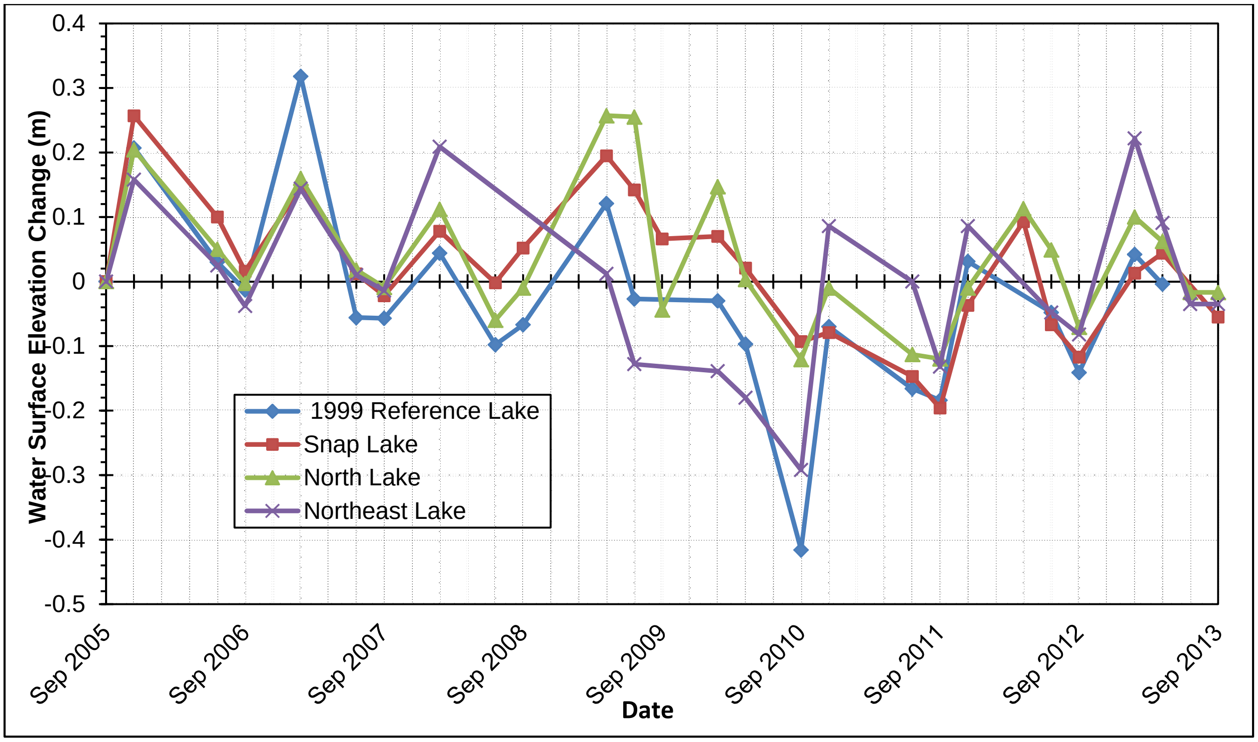
North Lake and Northeast Lake are hydraulically connected to Snap Lake, whereas 1999 Reference Lake is in a different drainage basin and is not hydraulically connected to Snap Lake, North Lake, or Northeast Lake. 1999 Reference Lake is used as an indicator of whether water elevation trends at Snap Lake, North Lake, and Northeast Lake are due to Mine effects or to regional environmental trends. The Environmental Assessment Report (De Beers 2002) predicted an increase of 5.3 centimetres (cm) in the mean water elevation of Snap Lake and decreases of 1.6 cm and 3.0 cm in the mean water elevations of Northeast Lake and North Lake respectively, as a result of the mining operations. However, to date, the predicted changes have been of a much lower magnitude than the annual variation due to environmental factors observed at all the study lakes; all four lakes have exhibited similar elevation trends.

The surveyed water elevations and the range of minimum and maximum water surface elevations between 2002 and 2013 for Snap Lake, 1999 Reference Lake, North Lake, and Northeast Lake are provided in Table 2-8. The water elevation of Snap Lake increased by approximately 0.062 m between 2012 and 2013 (Table 2-8, Figure 2-12), but remained within the range of elevations surveyed between 2002 and 2012.

2.4.3.1 Snap Lake

The discharges at the Snap Lake inflow (H4) station and outflow (H1 and H2) stations from 1999 to 2013 are shown in Figures 2-13 and 2-14. Inflows and outflows were within historical norms. Peak freshet during 2013 was not captured at the inflow station (H4), but occurred prior to May 27, 2013 (Figure 2-13). Peak outflow at outflow stations H1 and H2 occurred on June 20, 2013.

Figure 2-12 Water Surface Elevations of Snap Lake, 1999 Reference Lake, North Lake, and Northeast Lake, 2005 to 2013



m = metre; Sep = September.

Table 2-8 Surveyed Water Elevations for Snap Lake and Reference Lakes, 2002 to 2013

Year	Month	Snap Lake (masl)	1999 Reference Lake (masl)	North Lake (masl)	Northeast Lake (masl)
2002	Average[(a)]	444.297	440.841	439.839	433.074
2004	Average[(a)]	444.112	440.711	439.718	432.935
2005	Average[(a)]	444.151	440.776	439.766	432.972
2006	May	444.404	440.966	439.909	433.057
2006	August	444.247	440.789	439.755	432.924
2006	September	444.163	440.746	439.702	432.861
2007	June	444.293	441.077	439.865	433.043
2007	August	444.159	440.703	439.723	432.909
2007	September	444.125	440.702	439.696	432.885
2008	June	444.225	440.803	439.817	433.108
2008	August	444.145	440.661	439.645	n/a
2008	September	444.199	440.692	439.695	n/a
2009	July	444.342	440.880	439.962	432.911
2009	August	444.289	440.732	439.960	432.771
2009	September	444.213	n/a	439.661	n/a
2010	June	444.217	440.729	439.852	432.760
2010	July	444.168	440.662	439.708	432.719
2010	September	444.054	440.343	439.584	432.607
2011	May	444.068	440.689	439.695	432.985
2011	July/August	444.000	440.593	439.592	432.899
2011	September	443.951	440.575	439.585	432.767
2012	May	444.11	440.689	439.695	432.985
2012	July	444.24	n/a	439.818[(b)]	n/a
2012	August	444.08	440.502	439.754[(b)]	432.851
2012	September	444.03	440.427	439.634[(b)]	432.817[(b)]
2013	June	444.160	440.801	439.805	433.121
2013	July	444.191	440.755	439.767	432.990
2013	September	444.092	-	439.688	432.864
Year-on-year change, 2012 to 2013	September 2012 to September 2013	+0.062	+0.011[(c)]	+0.054	+0.047

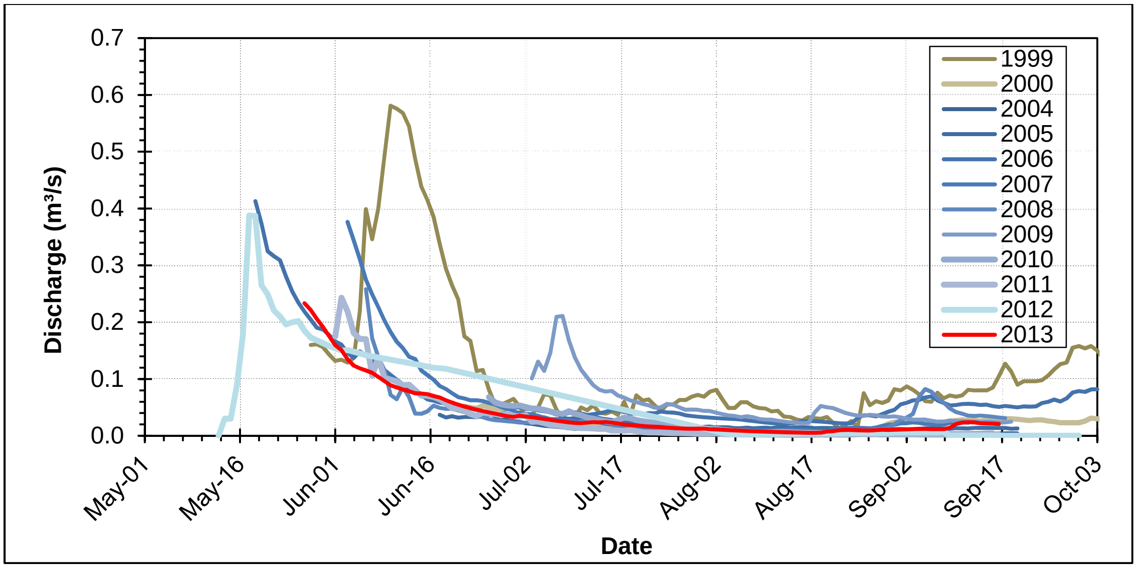
a) Average of the spring, summer, and fall surveyed water elevations.

b) Elevations calculated using stage-discharge rating curve and measured discharge flows since survey data were incorrect due to field procedure errors. Recommendations were made to De Beers in 2013 to reduce the possibility of errors.

c) Calculated from May 2012 to June 2013 since survey data missing for other comparison dates.

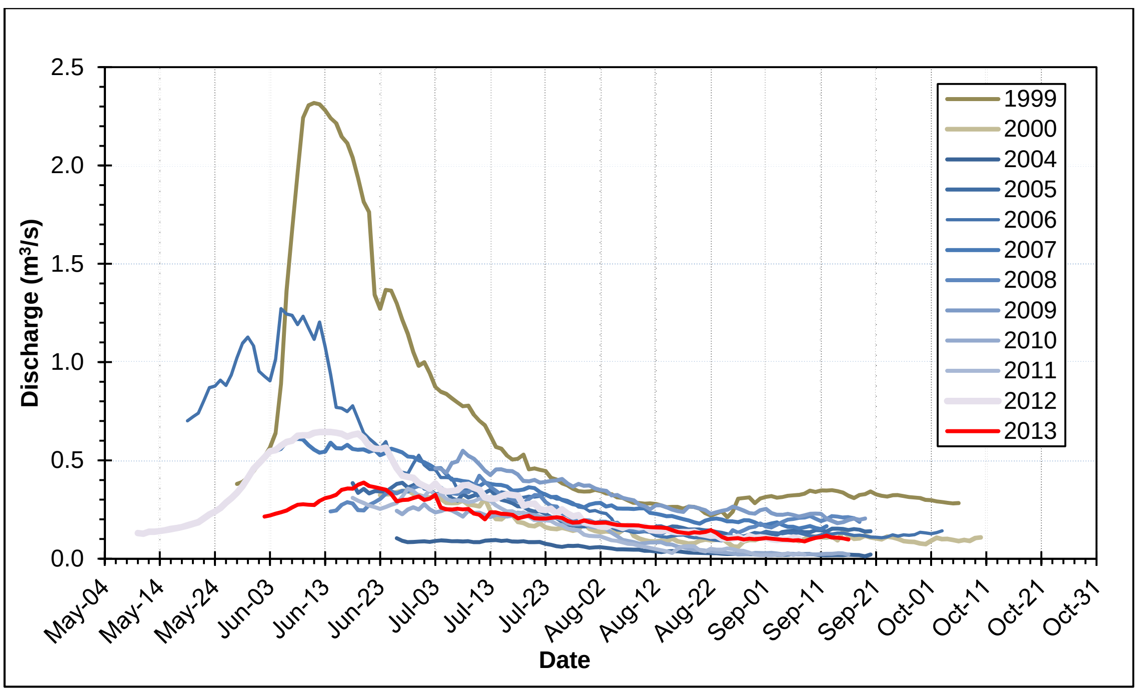
masl = metres above sea level; n/a = not available.

Figure 2-13 Discharge at Snap Lake Inflow (Station H4), 1999 to 2013



m^3/s = cubic metres per second; Jun = June; Jul = July; Aug = August; Sep = September; Oct = October.

Figure 2-14 Discharge at Snap Lake Outflow (Stations H1 and H2), 1999 to 2013



m^3/s = cubic metres per second; Jun = June; Jul = July; Aug = August; Sep = September; Oct = October.

Outflow discharge measurements for Snap Lake from 2001 to 2013 are provided in Table 2-9.

Table 2-9 Outflow Discharges for Snap Lake (Sum of Stations H1 and H2), 2001 to 2013

Date	Discharge [m^3/s]
May 29, 2001	0.598
June 9, 2002	0.415
August 12, 2002	0.365
October 1, 2002	0.250
June 26, 2004	0.174
September 21, 2004	0.043
June 18, 2005	0.410
September 20, 2005	0.145
May 19, 2006	0.658
August 3, 2006	0.279
October 3, 2006	0.189
June 3, 2007	0.516
August 15, 2007	0.277
September 12, 2007	0.202
June 9, 2008	0.313
August 13, 2008	0.115
September 18, 2008	0.164
July 2, 2009	0.481
August 24, 2009	0.258
September 19, 2009	0.220
June 23, 2010	0.211
July 31, 2010	0.182
September 16, 2010	0.035
May 28, 2011	0.142[a]
July 31, 2011	0.128
September 18, 2011	0.032
May 28, 2012	0.348
August 3, 2012	0.184
September 7, 2012	0.087
June 2, 2013	0.218
July 7, 2013	0.244
July 19, 2013	0.238
August 10, 2013	0.158
September 16, 2013	0.094

a) Flow through Station H2 not included due to ice blockage.

m^3/s = cubic metres per second.

2.4.3.2 1999 Reference Lake

Surveyed elevations and corresponding outflow discharge measurements for 1999 Reference Lake are provided in Table 2-10. The water elevation of 1999 Reference Lake increased by approximately 0.011 m between 2012 and 2013, and remained within the range of elevations surveyed between 2002 and 2012.

Table 2-10 Measured Water Elevation and Outflow Discharges for 1999 Reference Lake, 2002 to 2013

Date	Geodetic Elevation (masl)	Discharge (m^3/s)
July 7, 2002	440.839	0.423
August 11, 2002	440.846	0.340
September 30, 2002	440.839	0.311
June 27, 2004	440.770	0.160
September 21, 2004	440.652	0.060
June 18, 2005	440.869	0.667
August 25, 2005	440.699	0.086
September 19, 2005	440.759	0.199
May 20, 2006	440.966	1.443
August 3, 2006	440.789	0.250
October 2, 2006	440.746	0.138
June 2, 2007	441.077	0.815
August 14, 2007	440.703	0.191
September 12, 2007	440.702	0.131
June 9, 2008	440.803	0.691
August 13, 2008	440.661	0.073
September 17, 2008	440.692	0.103
July 2, 2009	440.880	0.925
August 17, 2009	440.732	0.178
September 9, 2009	n/a	0.129
June 24, 2010	440.729	0.193
July 31, 2010	440.662	0.080
September 15, 2010	440.343	0.012
May 28, 2011	440.689	0.290
August 1, 2011	440.593	0.033
September 18, 2011	440.575	0.023
May 28, 2012	440.689	0.302
August 5, 2012	440.502	0.138
September 8, 2012	440.427	0.055
June 3, 2013	440.801	0.409
July 18, 2013	440.755	0.244

masl = metres above sea level; m^3/s = cubic metres per second; n/a = not available.

2.4.3.3 North Lake

Surveyed water elevations and corresponding outflow discharges for North Lake are provided in Table 2-11. The water elevation of North Lake increased by approximately 0.054 m between 2012 and 2013, and remained within the range of elevations surveyed between 2002 and 2012.

Table 2-11 Measured Water Elevation and Outflow Discharges for North Lake, 2002 to 2013

Date	Geodetic Elevation (masl)	Discharge (m^3/s)
July 8, 2002	439.865	0.087
August 11, 2002	439.846	0.072
September 30, 2002	439.807	0.046
June 25, 2004	439.784	n/a
September 21, 2004	439.652	0.012
June 17, 2005	439.865	n/a
August 25, 2005	439.727	n/a
September 19, 2005	439.705	0.022
May 20, 2006	439.909	0.128
August 3, 2006	439.755	0.046
October 2, 2006	439.702	0.025
June 3, 2007	439.870	0.093
August 14, 2007	439.723	0.026
September 12, 2007	439.696	0.018
June 9, 2008	439.817	n/a
August 13, 2008	439.645	0.021
September 17, 2008	439.695	0.020
July 1, 2009	439.962	0.146
August 17, 2009	439.960	0.078
September 18, 2009	439.661	0.011
June 23, 2010	439.852	0.055
July 31, 2010	439.708	0.034
September 15, 2010	439.584	0.005
May 28, 2011	439.695	n/a
August 1, 2011	439.592	0.007
September 18, 2011	439.585	0.002
May 28, 2012	439.695	n/a
July 6, 2012	439.818[a]	0.052
August 5, 2012	439.754[a]	0.027
September 8, 2012	439.634[a]	0.008
June 2, 2013	439.805	n/a
July 18, 2013	439.767	0.059
September 16, 2013	439.688	0.007

a) Elevations calculated using stage-discharge rating curve and measured discharge flows since survey data were incorrect due to field procedure errors. Recommendations were made to De Beers in 2013 to reduce the possibility of errors.

masl = metres above sea level; m^3/s = cubic metres per second; n/a = not available.

2.4.3.4 Northeast Lake

Surveyed water surface elevations and corresponding outflow discharges for Northeast Lake are provided in Table 2-12. The water elevation of Northeast Lake increased by approximately 0.047 m between 2012 and 2013, and remained within the range of elevations surveyed between 2002 and 2012.

Table 2-12 Measured Water Elevations and Outflow Discharges for Northeast Lake, 2002 to 2013

Date	Geodetic Elevation (masl)	Discharge (m^3/s)
July 8, 2002	433.117	1.373
August 11, 2002	433.068	0.754
September 30, 2002	433.037	0.526
June 25, 2004	432.993	0.107
September 21, 2004	432.877	0.080
June 18, 2005	433.102	1.592
August 25, 2005	432.917	0.228
September 20, 2005	432.897	0.227
May 20, 2006	433.057	1.055
August 3, 2006	432.924	0.251
October 3, 2006	432.861	0.137
June 2, 2007	433.043	0.653
August 14, 2007	432.909	0.242
September 11, 2007	432.885	0.160
June 8, 2008	433.108	1.349
August 13, 2008	n/a	0.187
September 17, 2008	n/a	0.142
July 1, 2009	432.911	1.582
August 17, 2009	432.771	0.378
September 18, 2009	n/a	0.243
June 23, 2010	432.76	0.322
July 30, 2010	432.719	0.119
September 15, 2010	432.607	0.022
May 28, 2011	432.985	0.238
July 31, 2011	432.899	0.035
September 18, 2011	432.767	0.041
May 28, 2012	432.985	0.241
August 5, 2012	432.851	0.232
September 8, 2012	432.817[a]	0.057
June 2, 2013	433.121	0.197[b]
July 18, 2013	432.99	1.001
September 16, 2013	432.864	0.105

a) Elevations calculated using stage-discharge rating curve and measured discharge flows since survey data were incorrect due to field procedure errors. Recommendations were made to De Beers in 2013 to reduce the possibility of errors .

b) Discharge value likely not correct – should be approximately one order of magnitude higher based on elevation survey.

masl = metres above sea level; m^3/s = cubic metres per second; n/a = not available.

2.4.3.5 Hydrology Summary

Streamflows and water elevations for Snap Lake, North Lake, 1999 Reference Lake, and Northeast Lake during 2013 were within values recorded between 1999 and 2012 and are considered within normal ranges. Water elevations for Snap Lake, North Lake, and Northeast Lake increased between 0.011 m and 0.062 m between 2012 and 2013, indicating that Snap Lake is following regional trends, and the effect of the Mine on the water elevation of Snap Lake remains low. Precipitation and evaporation at Snap Lake during 2013 were also within normal historical ranges.

2.4.4 Seasonal Water Temperature

Water temperature data collected between July and September 2013 from SLMB, SLNW, Northeast Lake, and Lake 13 are presented in Figures 2-7 to 2-11. Water temperature data collected from DSL1, DSL2, and LCB are presented in the Downstream Lakes Special Study (Chapter 11, Section 2).

2.4.4.1 Shallow Habitat Temperature

The shallow habitat temperature data collected in SLMB, SLNW, Northeast Lake, and Lake 13 followed similar trends through the late spring and summer (Figure 2-15). In SLMB, SLNW, and Lake 13, water temperature increased sharply from July 6 to 9, 2013 to a maximum of 18°C and then declined to a minimum of 12°C within the following eight days. This peak was not captured in Northeast Lake because the thermographs were installed in this lake on July 11, 2013.

In SLMB, SLNW, Northeast Lake, and Lake 13, water temperature steadily increased from mid-July to mid-August to a maximum of 20°C, and decreased gradually until mid-September to a minimum of 10°C.

2.4.4.2 Deep Basins Temperature

Overall, the deep basin temperature measured in SLMB, SLNW, Northeast Lake, and Lake 13 (Figures 2-16 to 2-19, Appendix 2B, Tables 2B-2 to 2B-5) show that:

- temperature decreased with lake depth;
- maximum temperatures were recorded at the surface of each lake; and,
- minimum temperatures were recorded closer to the substrate.

Similar to the temperature trends observed in the shallow habitat, the thermographs installed at the surface in the deep basins increased rapidly at the beginning of July and reached a first peak on July 11, and then decreased over a period of approximately ten days. This peak was not captured in Northeast Lake because the thermographs were installed in this lake on July 11, 2013. For the rest of the summer, temperatures recorded closer to the surface of SLMB, SLNW, Lake 13, and Northeast Lake also followed similar trends, increasing until mid-August to a second peak showing maximum temperatures close to

20°C, and decreasing to minimum temperatures close to 11°C in early September (Figures 2-16 to 2-19, Appendix 2B, Tables 2B-2 to 2B-5).

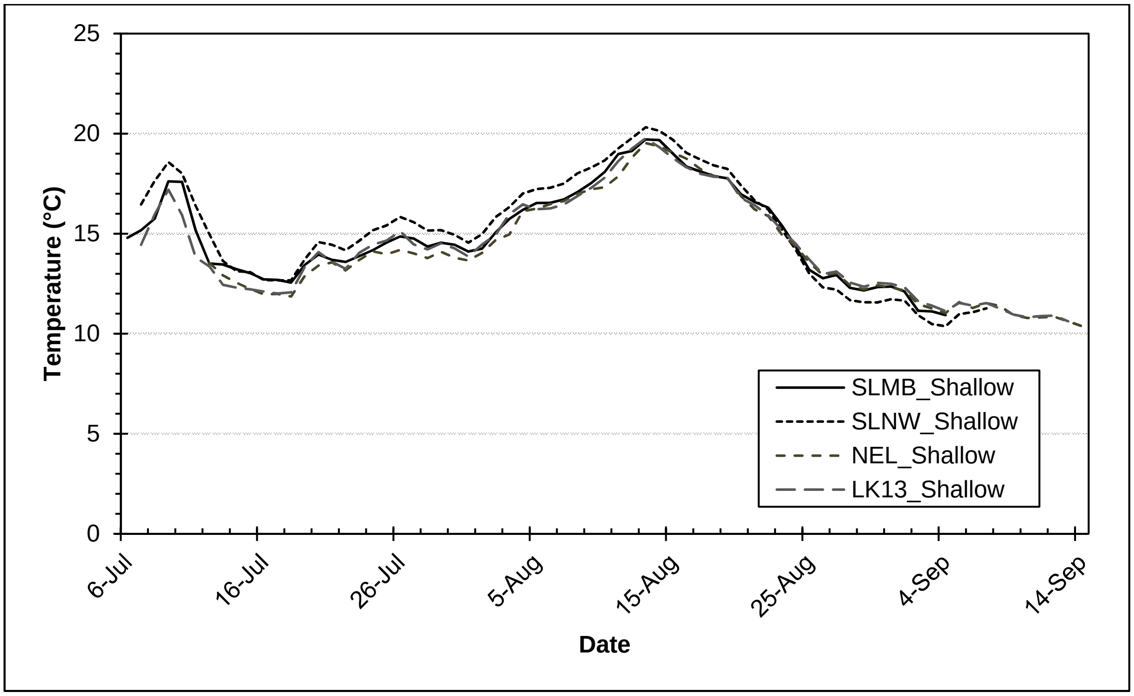
On July 12, 2013, the water temperature at SLMB at 19.3 m and 22.3 m increased very rapidly (Figure 2-16). At 10:45 AM, water temperature at both depths was approximately 5°C; by 1:25 PM, temperature had risen to 12.8°C at 19.3 m but remained at approximately 5°C at 22.3 m. By 11:25 PM, water temperature at 19.3 m and 22.3 m was 14°C and 11°C, respectively, and had reached the temperature recorded by the loggers installed closer to the surface. Similar rapid increases in temperature were also observed on the same day at SLNW (Figure 2-17) at 9.3 m, in Northeast Lake at 17.5 m and 20.5 m (Figure 2-18), and in Lake 13 at 12.3 m and 15.3 m (Figure 2-19). This abrupt change in temperature was likely caused by high winds (Figure 2-5) mixing the water column.

Temperatures near the bottom of SLMB (depth of 31.3 m) and Northeast Lake (depth of 26.5 m) were stable at 3.5°C and 5°C respectively until the end of July; the temperature in both lakes steadily increased to 8°C until retrieval of the thermographs in September 2013 (Figures 2-16 and 2-18, Appendix 2B, Tables 2B-2 and 2B-4). In Lake 13, temperature at the bottom of the lake (depth of 18.3 m) followed a similar trend as the temperature recorded throughout the water column, but with less temperature fluctuations (Figure 2-19, Appendix 2B, Table 2B-5).

At SLNW, temperature fluctuations throughout the summer were only observed to a maximum depth of 12.3 m (Figure 2-17). Temperature at the 12.3 m depth at SLNW was stable at approximately 5°C until the end of July, and then increased rapidly to 10°C in early September and remained at this temperature until the time of thermograph retrieval (Figure 2-17, Appendix 2B, Table 2B-3). Temperature loggers installed between 15.3 and 39.3 m at SLNW recorded steady temperatures throughout the summer that ranged from an average of 4.5°C at 15.3 m to 3.4°C at 39.3 m (Figure 2-17, Appendix 2B, Table 2B-3). Temperatures recorded by the thermographs are consistent with temperatures collected during the monthly water quality programs.

The minimum, maximum, and mean temperatures from temperature logger data recorded over the entire monitoring program (i.e., all depths, all days included) for each station are presented in Appendix 2B, Figure 2B-1. Maximum (i.e., surface) temperatures recorded for SLMB, SLNW, and Lake 13 were similar (i.e., close to 20°C), but lower for Northeast Lake (i.e., 18.5°C), likely because the surface temperature logger at Northeast Lake was installed at 2.5 m from the surface, and not at 0.3 m as for the other lakes (Table 2-3). Minimum temperatures recorded near the bottom in SLMB and SLNW were similar (close to 4°C), but minimum temperatures were higher in Northeast Lake and Lake 13, likely due to shallower depths. Mean temperatures are variable due to a differing number of measurements at various depths for the lakes. Differences in mean temperatures between SLMB and SLNW are likely due to wind-induced mixing within the main basin of the lake. There is also the potential that some of the mixing observed may be due to the presence of the diffuser in SLMB. However, in the 2012 plume characterization study (Golder 2013), mixing patterns close to the diffuser discharge site were observed, but water turbulence due to the diffuser discharge could not be observed farther than 70 m from the diffuser pipe. As the temperature loggers were installed at approximately 200 m from the diffuser pipe, it is unlikely the diffuser pipe was the main source of water column mixing observed in Figure 2-15.

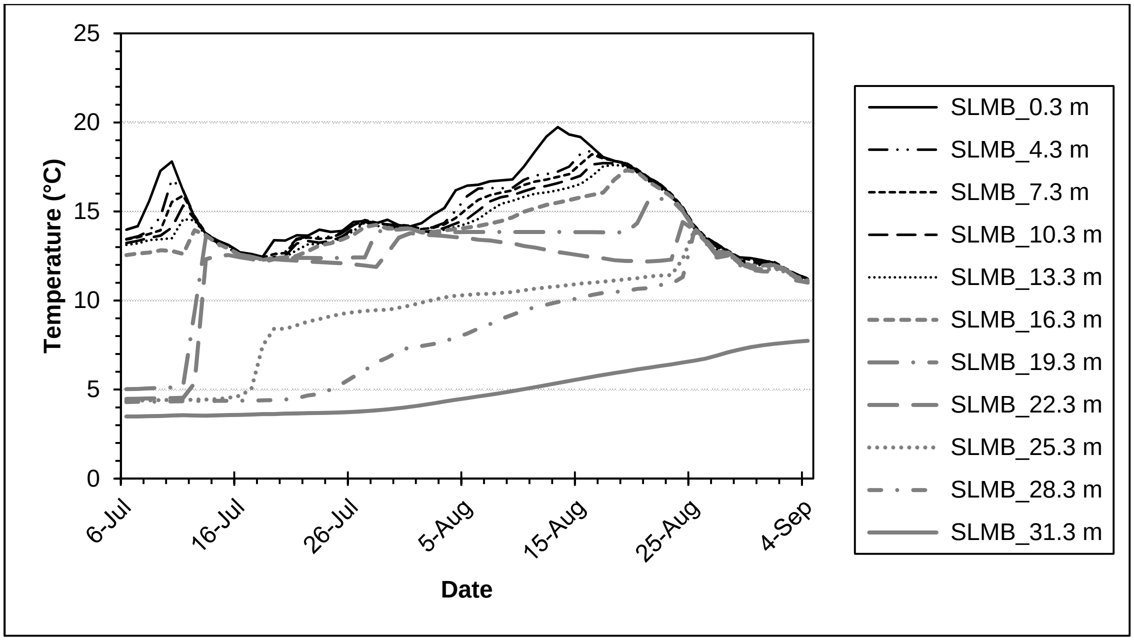
Figure 2-15 Shallow Sites Temperature in Snap Lake, Northeast Lake and Lake 13, Summer 2013



Note: Data represent temperature average of two shallow sites. Temperature loggers were installed at 0.5 m from bottom. Total depth ranged from 0.9 to 1.2 m.

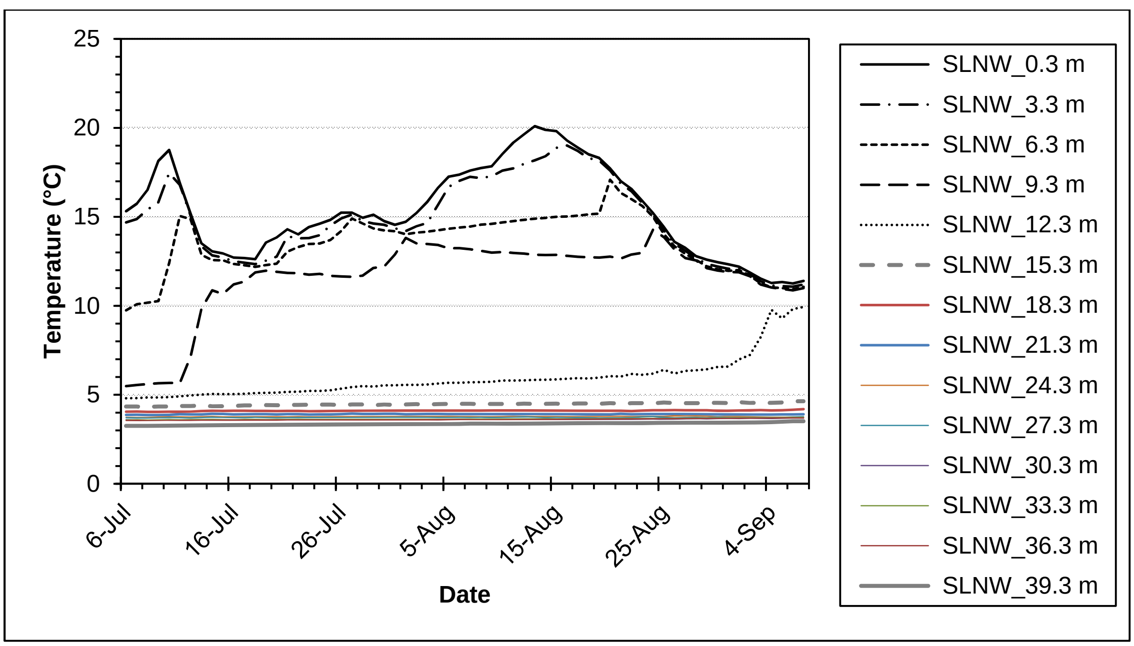
ºC = degrees Celsius; SLMB = Snap Lake Main Basin; SLNW = Snap Lake Northwest Arm; NEL = Northeast Lake; LK13 = Lake 13; Jul = July; Aug = August; Sep = September.

Figure 2-16 Snap Lake Main Basin Deep Site Water Temperature from Surface of Lake, Summer 2013



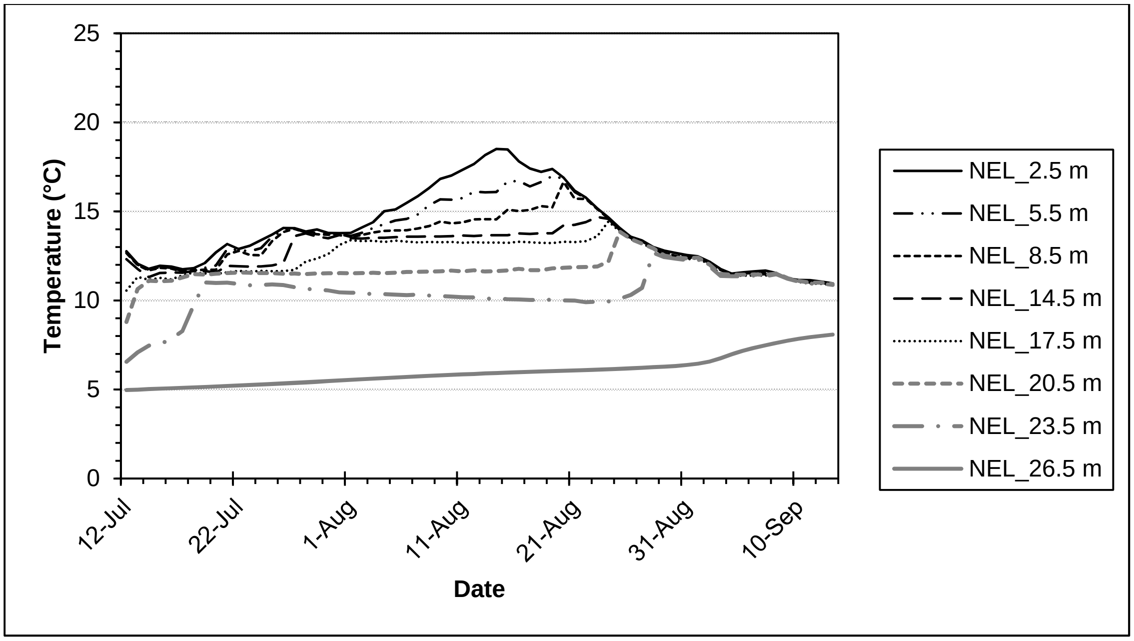
Note: Total water depth = 31.8 m.
ºC = degrees Celsius; m = metre; SLMB = Snap Lake Main Basin; Jul = July; Aug = August; Sep = September.

Figure 2-17 Snap Lake Northwest Arm Deep Site Water Temperature from Surface of Lake, Summer 2013



Note: Total water depth = 39.8 m.
ºC = degrees Celsius; m = metre; SLNW = Snap Lake northwest arm; Jul = July; Aug = August; Sep = September.

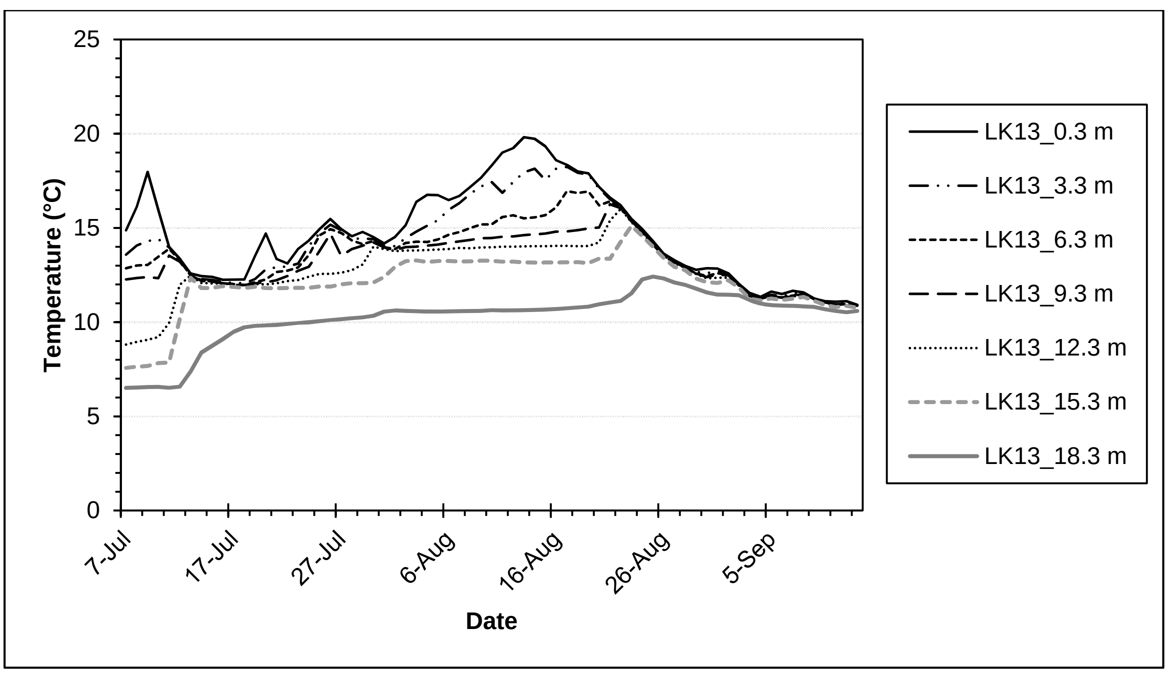
Figure 2-18 Northeast Lake Deep Site Water Temperature from Surface of Lake, Summer 2013



Note: Temperature logger installed at 11.5 m was damaged. No probe was installed at 0.3 m from the surface. Total water depth = 27 m.

ºC = degrees Celsius; m = metre; NEL = Northeast Lake; Jul = July; Aug = August; Sep = September.

Figure 2-19 Lake 13 Deep Site Water Temperature from Surface of Lake, Summer 2013



Note: Surface reading (0.3 m) is the average of readings of two temperature loggers installed at the same depth. Total water depth = 18.8 m.

ºC = degrees Celsius; m = metre; LK13 = Lake 13; Jul = July; Aug = August; Sep = September.

2.4.5 Ice Thickness and Ice Cover Monitoring

2.4.5.1 Ice Thickness

Ice thickness measured in Snap Lake from 2005 to 2013 is presented in Figure 2-20. Measurements collected in Northeast Lake from 2008 to 2013, and in Lake 13 in May 2013 are presented in Figure 2-21. Ice thickness was higher in 2008 and 2012 for both lakes than other years sampled.

Monthly and annual mean thicknesses for Snap Lake and Northeast Lake, as well as ice thickness measured in Lake 13 in May 2013 are presented by year in Figures 2-22 and 2-23. In general, ice thickness measurements from 2013 are in the range of measurements from other years sampled. Ice thickness seems to grow from January to April and stabilize from April to June (Figure 2-22). Ice appears to be thicker in Northeast Lake than in Snap Lake (Figures 2-22 and 2-23); however, ice thickness measurements in Northeast Lake were not done as frequently as those for Snap Lake, and were mostly collected in February and April (Figure 2-21), versus every winter month in Snap Lake.

2.4.5.2 Ice Cover and Open-Water Days

The ice-off and ice-on dates observed by De Beers Environmental staff at Snap Lake from 2008 to 2013 are summarized in Table 2-10. The total days of ice cover in 2013 was 231 days, consistent with previous years (Table 2-13).

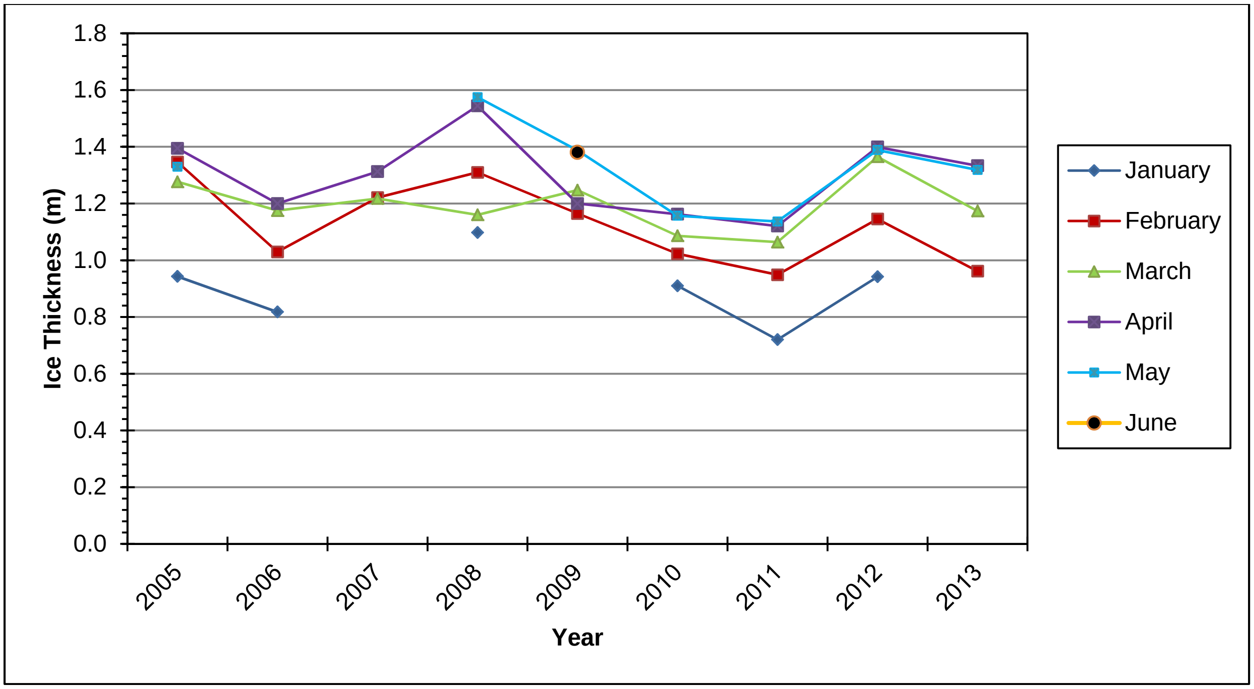
Table 2-13 Days of Ice Cover Versus Open-Water for Snap Lake, 2008 to 2012

Year	Ice-Off Date	Ice-On Date	Days of Ice Cover	Days of Open-water
2008	June 6, 2008	October 24, 2008	226	140
2009	June 7, 2009	October 12, 2009	237	128
2010	June 14, 2010	October 16, 2010	240	125
2011	June 17, 2011	October 28, 2011	231	134
2012	June 10, 2012	October 27, 2012	226	140
2013[a]	June 16, 2013	October 28, 2013	231	134

Note: Ice-Off Date = Last observation of ice on main basin of Snap Lake; Ice-On Date = Observation of main basin of Snap Lake covered by ice.

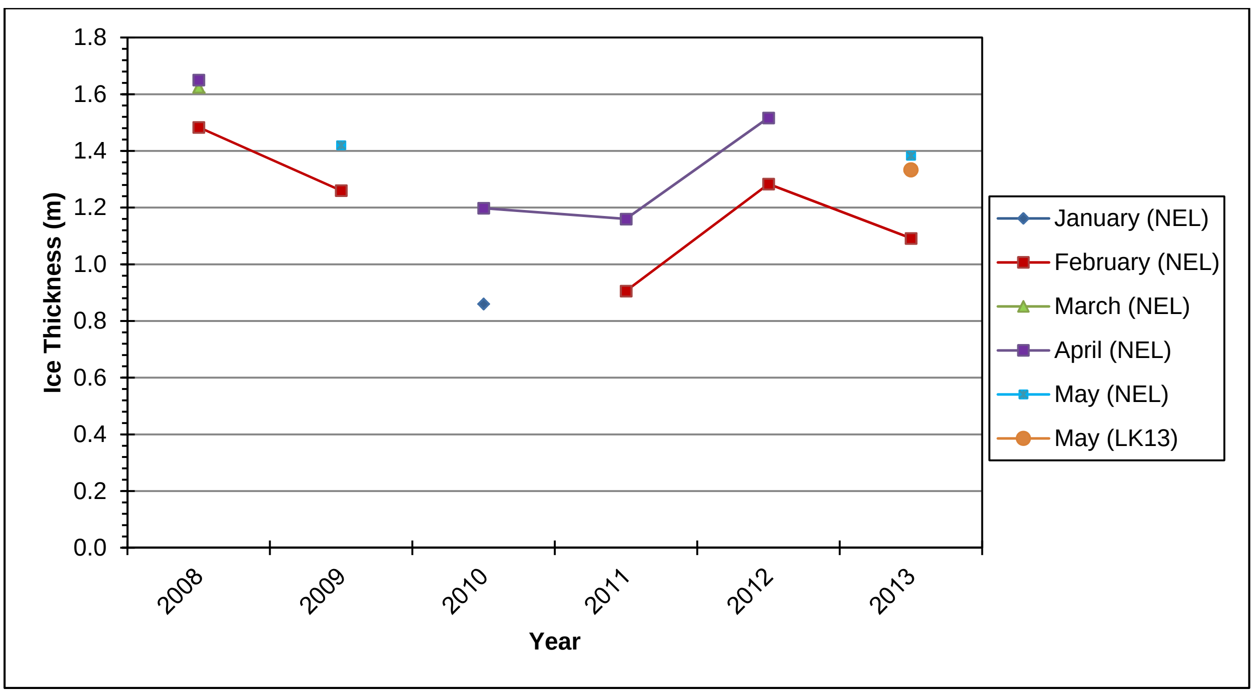
a) Ice-Off and Ice-On dates observed in the west arm of Snap Lake, not the main basin.

Figure 2-20 Monthly Ice Thickness in Snap Lake, 2005 to 2013



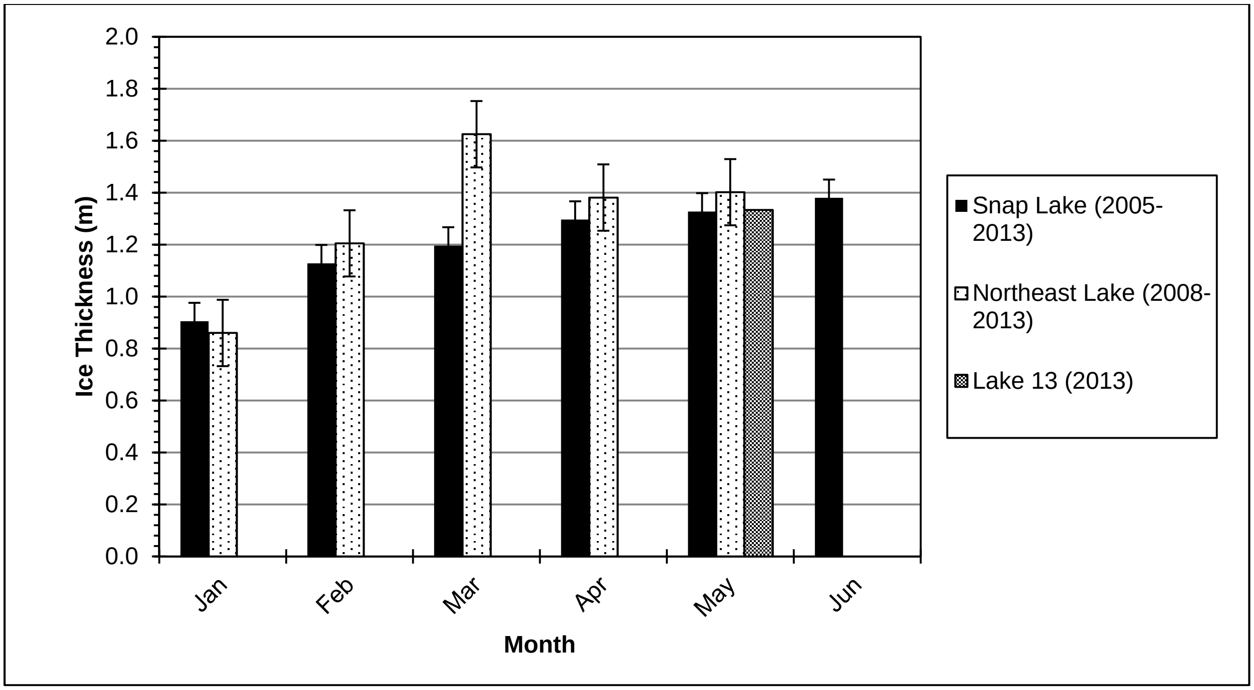
m = metre.

Figure 2-21 Monthly Ice Thickness in Northeast Lake (2008 to 2013) and Lake 13 (2013)



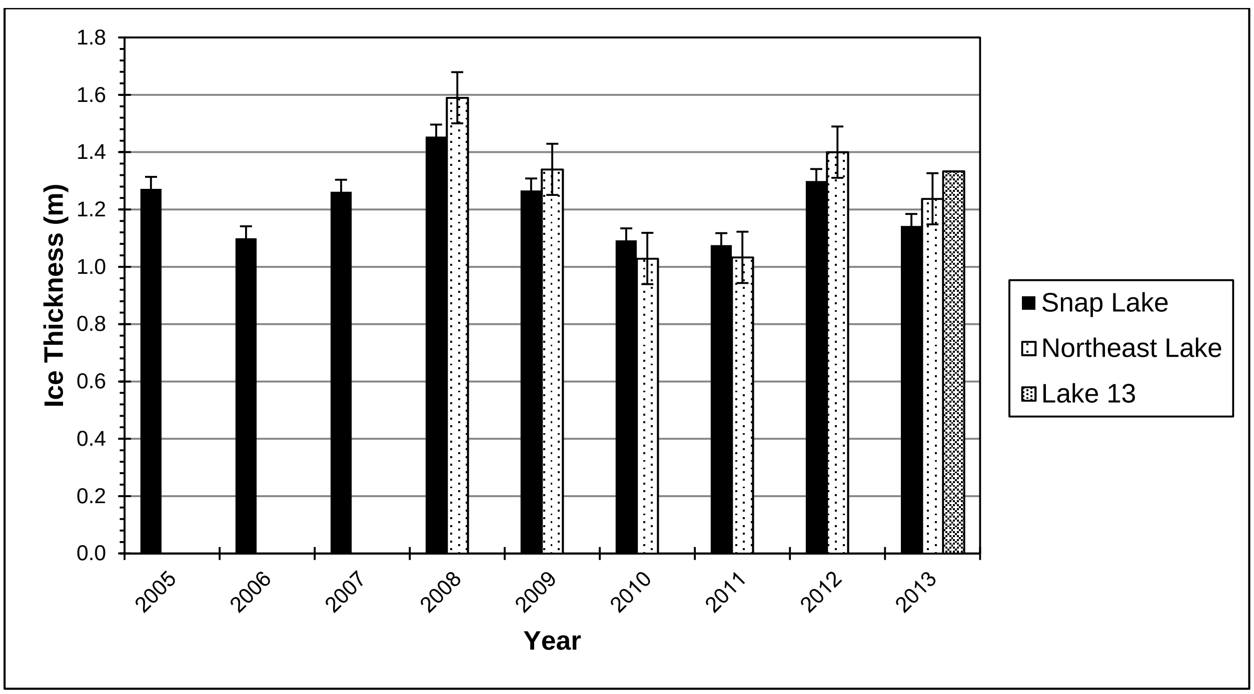
m = metre; NEL = northeast Lake; LK13 = Lake 13.

Figure 2-22 Average Monthly Ice Thickness in Snap Lake and Northeast Lake



m = metre.

Figure 2-23 Average Yearly Ice Thickness in Snap Lake and Northeast Lake



m = metre.

2.5 Conclusions

2.5.1 Key Question 1: What are the General Conditions of the Mine Site and the Local Environment under which the AEMP is Conducted, Independent of Mining-related Activities and Considering Unanticipated Mining Events such as Spills?

One reportable spill occurred close (i.e., less than 50 m) to Snap Lake; however, this spill did not enter Snap Lake, and is not anticipated to have a measurable effect on the aquatic environment.

Seepage at the Mine in 2013 is not anticipated to have a measurable effect on the aquatic environment.

The Mine has undergone operational changes from the original project description or from last year.

For example:

- a sedimentation / floc tank has been added upstream of the WMP to pre-treat runoff from the North Pile (new to this year).

Treated effluent from the WTP is discharged daily into Snap Lake through minewater outlet pipelines equipped with diffusers. One permanent diffuser discharged water all throughout 2013. A second diffuser was permanently installed in fall 2013 and started discharging treated effluent on October 6, 2013; a temporary diffuser was used from May 18, 2013, to October 5, 2013. A total of 13.7 Mm^3 of treated effluent was discharged in 2013, which is an increase of 28% over the 10.7 Mm^3 discharged in 2012. Modifications to the existing treated effluent discharge system increased the daily treated effluent discharge volume by approximately 8,000 m^3/day in 2013 compared to 2012. The minimum and maximum discharges occurred in July and in May 2013, respectively.

The total annual rainfall recorded at the Hill Station for Snap Lake in 2013 was 143.8 mm, which is approximately 3.6% higher than in 2012 (138.7 mm), 73.6% higher than the Yellowknife total for 2013 (83.8 mm), and 15.8% lower than the Yellowknife long-term (1981 to 2010) annual precipitation average of 170.8 mm. Monthly rainfall followed the same pattern observed in Yellowknife from 1981 to 2010.

Average annual air temperature at the Hill and Lake Stations were respectively -8.21°C and -8.35°C, which is approximately 1.5°C colder than in 2012, and the average annual temperature measured at Snap Lake (including Hill and Lake Stations) from 2003 to 2012. It was also 4.0°C colder than the annual temperature of -4.3°C for Yellowknife during 1981 to 2010. As expected, Lake Station solar radiation was negative from January to April and from October to December 2013.

Similar to previous years, predominant winds at the Mine in 2013 were from the east and east-southeast. Lower wind speeds were measured from the south-west.

The water elevation of Snap Lake increased by approximately 0.062 m between 2012 and 2013 and remained within the range of elevations surveyed between 2002 and 2012. Snap Lake had a lower range of elevation changes between 2002 and 2013 than the 1999 Reference Lake, indicating that the Mine operations likely had a minimal effect on fluctuations in the Snap Lake water surface elevation.

2.5.2 Key Question 2: Is there a Habitat Difference between Snap Lake and the Reference Lakes in terms of Seasonal Water Temperature and Ice-cover?

The lakes showed a similar pattern in temperatures, with a temperatures in the upper layers increasing from mid-July to mid-August, and then decreasing into September. Thermoclines were observed at the deep locations in late summer. Maximum temperatures were recorded at the surface for each lake; SLMB, SLNW, Lake 13, and Northeast Lake were similar in surface temperatures. Minimum temperatures were recorded near the bottom of each lake, with minimum temperatures similar for SLMB and SLNW, but slightly higher for Northeast Lake and Lake 13, likely due to shallower depths of the lakes. Differences in mean temperatures between SLMB and SLNW are likely due to wind-induced mixing within the main basin of the lake.

Ice thickness measurements from 2013 are in the range of measurements from other years sampled. In general, ice seems to be thicker in Northeast Lake than in Snap Lake. However, Northeast Lake ice thickness measurements have not been collected as frequently those for as Snap Lake. Snap Lake had 231 days of ice cover in 2013, which is similar to the past five years.

2.6 Recommendations

Based on the results, the following change to the 2014 AEMP program is recommended:

- Year-to-year changes to the Mine, which have the potential to affect the environment should also be reviewed and considered.
- The temperature logger program should be implemented earlier in the year, if possible, to capture variations in spring temperatures.

2.7 References

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SECTION 3

WATER QUALITY

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LIST OF ACRONYMS

Term	Definition
AB	Alberta
AEMP	Aquatic Effects Monitoring Program
ALS	ALS Canada Ltd.
AML	average monthly limit
AO	aesthetic objectives
APHA	American Public Health Association
BC	British Columbia
BOD	biochemical oxygen demand
$CaCO_3$	calcium carbonate
CCME	Canadian Council of Ministers of the Environment
CCMS	collision cell inductively coupled plasma mass spectrometry
De Beers	De Beers Canada Inc.
DDW	distilled de-ionized water
DFO	Fisheries and Oceans Canada
DO	dissolved oxygen
DSL	downstream lake
DSL1	Downstream Lake 1
DSL2	Downstream Lake 2
EAR	Environmental Assessment Report
E. coli	*Escherichia coli*
EEM	Environmental Effects Monitoring
e.g.	for example
F1	F1 hydrocarbon fractions
F2	F2 hydrocarbon fractions
Flett	Flett Research Limited
GPS	global positioning system
H1	Hydrology Station 1
H2	Hydrology Station 2
HydroQual	HydroQual Laboratories
i.e.	that is
IL	inland lake
LC	lethal concentration
LK13	Lake 13
Main Basin	Main Basin of Snap Lake
Max Grab	maximum allowable concentrations in any grab sample
MB	Manitoba
MDS	Multi-parameter Display System
Mine	Snap Lake Mine
MAC	maximum acceptable concentrations
MVLWB	Mackenzie Valley Land and Water Board
n	sample count
N	nitrogen

Term	Definition
NEL	Northeast Lake
NWT	Northwest Territories
P	probability
PVC	polyvinyl chloride
QA	quality assurance
QC	quality control
QS	Quick Sample
r	Pearson's correlation co-efficient
RPD	relative percent difference
SD	standard deviation
SNAP	Snap Lake
SNP	Surveillance Network Program
SSWQO	site-specific water quality objective
TDS	total dissolved solids
TKN	total Kjeldahl nitrogen
TN	total nitrogen
TP	total phosphorus
TSS	total suspended solids
TWTP	temporary water treatment plant
UofA	University of Alberta Biogeochemical Analytical Service Laboratory
UTM	Universal Transverse Mercator
WQG	water quality guideline
WOE	weight of evidence
WTP	water treatment plant
↑	increase
↓	decrease

LIST OF SYMBOLS

Term	Definition
C_b	background lake concentration in Snap Lake, represented by the average TDS concentrations from main basin stations : SNAP03, SNAP05, and SNAP06
C_d	maximum TDS concentration at the three diffuser stations SNP 02-20d, SNP 02 20e, and SNP 02-20f
C_e	combined flow-weighted average TDS concentration in the treated effluent from SNP 02-17B
C_{WTPi}	concentration in the treated effluent from SNP 02-17B during sampling event i
DF	minimum dilution factor of the diffuser(s)
FWC_{WTP}	flow-weighted average TP concentration in the treated effluent from the WTP (SNP 02-17B), which includes effluent from the TWTP
F_{WTPi}	daily discharge volume at SNP 02-17B associated with sampling event i
GM $\bar{y}$	geometric mean
i	sampling event
n	number of samples
V_{WTP}	total volume of discharge at SNP 02-17B
y	bacterial counts

UNITS OF MEASURE

Term	Definition
%	percent
>	greater than
±	plus or minus
°C	degrees Celsius
µeq/L	microequivalent per litre
µm	micrometre
µS/cm	microSiemens per centimetre
cm	centimetre
CFU/100 mL	colony forming unit per 100 millilitres
kg	kilogram
kg/y	kilograms per year
km	kilometre
km^2	square kilometres
L	litre
m	metre
m^3/d	cubic metres per day
meq/L	milliequivalent per litre
mg/L	milligrams per litre
mg-N/L	milligrams as nitrogen per litre
mg-P/L	milligrams as phosphorus per litre
mL	millilitre
Mm^3	million cubic metres
MPN/100mL	most probable number per 100 millilitres

3 WATER QUALITY

3.1 Introduction

3.1.1 Background

3.1.1.1 Snap Lake

Baseline water quality data were collected from 1998 to 2001 in Snap Lake as part of the work completed to support the Environmental Assessment Report (EAR; De Beers 2002). Additional water quality data were collected in Snap Lake in 2002 and 2003 during the Care and Maintenance phase of the Snap Lake Mine (Mine) before construction began. Water quality monitoring in Snap Lake under the Aquatic Effects Monitoring Program (AEMP) for the Mine began in May 2004. Discharge of treated effluent to Snap Lake, which refers to effluent after removal of suspended solids and pH adjustment at the Water Treatment Plant (WTP), began on June 22, 2004 using a temporary diffuser.

In 2013, De Beers conducted AEMP water quality monitoring to comply with Water Licence MV2011L2-0004 (MVLWB 2012, 2013a), effective June 14, 2012 and the Aquatic Effects Monitoring Program Design Plan (hereafter referred to as the 2005 AEMP Design Plan) (De Beers 2005a) until the 2013 AEMP Design Plan was approved and implemented in April 2013 (De Beers 2014a). The water quality component of the AEMP was updated in the 2013 AEMP Design Plan to account for changes in Snap Lake water quality that have occurred since the Mine began operating in 2004. The focus of water quality monitoring in the 2013 AEMP Design Plan has shifted from spatial gradients within Snap Lake, which have lessened over time, to temporal trends in Snap Lake, comparison to reference lakes, and changes downstream of Snap Lake.

3.1.1.2 Reference Lakes: Northeast Lake and Lake 13

Northeast Lake

Northeast Lake is located 10 km northeast of Snap Lake and is also a relatively small lake, with a surface area of approximately 18 km^2 (Section 1, Figure 1-2). In 2006, Northeast Lake was selected as a reference lake. Under the Environment Canada Environmental Effects Monitoring (EEM) program, a reference area is defined as waters frequented by fish that are not exposed to treated effluent, with fish and fish habitat as similar as possible to the exposure area (Environment Canada 2012).

Northeast Lake was selected as the reference lake using a two-step process. First, a desktop screening analysis short-listed six possible reference lakes from 26 candidate lakes (Golder 2005a). Field surveys were then completed in each of the six short-listed lakes and results were compared to Snap Lake monitoring data (Golder 2005b). Northeast Lake was selected as an appropriate reference lake based on its similarity to Snap Lake in terms of bathymetry, water quality, sediment quality, and fish community composition. Stakeholder input was considered during the lake selection process. The Mackenzie Valley

Land and Water Board (MVLWB) provided final approval to accept Northeast Lake as the reference lake as a condition of the De Beers Snap Lake Mine Water Licence in April 2006 (MVLWB 2006).

Water quality monitoring started at Northeast Lake as a component of the AEMP in July 2006. The purpose of collecting water quality data at Northeast Lake is to help separate natural variability and background environmental changes, such as effects of climate change, from potential effects on Snap Lake resulting from the Mine. Historical data from Northeast Lake are available for 2002, 2004, and 2005.

Lake 13

In 2012, the Aquatic Effects Re-evaluation Report recommended the addition of a second reference lake (De Beers 2012a) to provide additional information regarding the regional context for Snap Lake. Due to the inherent natural differences in lakes within the region, particularly nutrient concentrations, a multiple reference lake design was recommended.

Lake 13, which is located 20 km northwest of Snap Lake (Section 1, Figure 1-2) and has a surface area of 11 km^2, was proposed as the second reference lake (De Beers 2012a). The recommendation of Lake 13 as the second reference lake was based on the results of the 2005 review of reference lakes, which indicated that Lake 13 was the second most similar lake to Snap Lake, after Northeast Lake (Golder 2005b) (Northeast Lake in Section 3.1.1.2). The MVLWB approved Lake 13 as the second reference lake, conditional on additional monitoring to assess the influence of the construction and operation of the winter access road on Lake 13 (MVLWB 2013b). The additional monitoring began in 2014, as per the Lake 13 Winter Road Monitoring Design Plan (De Beers 2014b); results from this monitoring will be included in Surveillance Network Program (SNP) reports in 2014.

The first year of routine AEMP monitoring of water quality in Lake 13 was 2013; however, historical water quality data are available from 2005, when the potential reference lakes were evaluated, and in 2012, when the AEMP included Lake 13 as a potential reference lake based on recommendations from the Aquatic Effects Re-evaluation Report (De Beers 2012a).

3.1.1.3 Objectives

The primary objectives of the water quality component of the AEMP, as defined in the 2013 AEMP Design Plan (De Beers 2014a), are to:

- characterize and interpret water quality in Snap Lake for the purpose of identifying any Project-related effects;
- verify and update the EAR predictions (De Beers 2002);
- support and inform management decisions made by Mine personnel (i.e., the Response Framework); and,
- recommend any necessary and appropriate changes to the water quality component of the AEMP for future years (De Beers 2014a).

Key Questions

To meet the primary objectives of the AEMP water quality component, analyses and interpretation of water quality data were focused on answering the following six key questions:

1. Are concentrations or loads of key water quality parameters in discharges to Snap Lake consistent with EAR predictions and below Water Licence limits?
2. Are concentrations of key water quality parameters in Snap Lake below AEMP benchmarks[1], and Water Licence limits?
3. Which water quality parameters are increasing over time in Snap Lake and nearby waterbodies, and how do concentrations of these parameters compare to AEMP benchmarks, concentrations in reference lakes, EAR predictions, and subsequent modelling predictions?
4. Are spatial and seasonal patterns in water quality in Snap Lake and downstream waterbodies consistent with predictions presented in the EAR and subsequent modelling predictions?
5. Is there evidence of acidification effects from the Mine on nearby waterbodies?
6. Is water from Snap Lake safe to drink?

In addition to answering the six key questions, the information used to answer these six key questions was integrated into the action level and weight of evidence (WOE) assessments.

The field survey and data analysis methods used to answer the key questions and assess action levels are described in Section 3.2. A summary of the quality assurance (QA) and quality control (QC) assessment on the 2013 data is provided in Section 3.3, followed by the 2013 results and conclusions (organized by key question and action level assessment), which are provided in Sections 3.4 and 3.5, respectively. Details of the QA/QC assessment, field profile data and laboratory results obtained during the 2013 AEMP, as well as additional temporal and spatial plots of parameters not presented in the main report are provided in the appendices:

- Appendix 3A Quality Assurance and Quality Control Procedures and Results for the Water Quality Program, which includes detailed methods and findings from the QA/QC assessment for the 2013 AEMP water quality program;
- Appendix 3B Nutrient Assessment, which describes the 2013 study completed as a follow-up to the 2012 Nutrient Assessment;
- Appendix 3C Field Data for 1999-2013, which includes field data collected in Snap Lake, the reference lakes, inland lakes, Streams S1 and S27, and King Lake;
- Appendix 3D Laboratory Water Quality Data in 2013, which provides laboratory results for Snap Lake, the reference lakes, inland lakes, Streams S1 and S27, and King Lake collected during the 2013 AEMP water quality program;

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[1] AEMP benchmarks are defined as either or both generic aquatic life guidelines (CCME 1999) and/or site-specific EAR benchmarks (De Beers 2002; Section 3.4.3).

- Appendix 3E Temporal Trends in Treated Effluent, which provides the laboratory results (2013) and temporal concentration and loading plots for treated effluent at SNP 02-17B (Key Question 1);
- Appendix 3F Toxicity, which provides details of toxicity test results in the water samples collected in treated effluent and in the edge of mixing zone in 2013 (Key Question 1 and 2);
- Appendix 3G Temporal Patterns, which provides additional results from the temporal assessment for Snap Lake, Northeast Lake and Lake 13 in 2013 (Key Question 3);
- Appendix 3H Spatial and Seasonal Patterns, which provides additional results from the spatial and seasonal assessment for Snap Lake, Northeast Lake and Lake 13 in 2013 (Key Question 4); and,
- Appendix 3I Water Quality Data for the Water Intake SNP 02-15; laboratory results for the water intake sampling location, as part of SNP from November 2012 to October 2013.

3.1.1.4 Supplemental Studies

In addition to the core AEMP program, special studies occur as necessary, and include research or other activities that support effects monitoring. These studies do not necessarily assess changes that may be related to the Mine, but rather focus on development of monitoring methods, further investigation of monitoring findings, or to fill data gaps. In 2013, two supplemental studies, the Downstream Lakes Special Study (Section 11.3) and the Nutrient Assessment (Appendix 3B) both had water quality components. The purpose of the Downstream Lakes Special Study, which included collecting water quality profile measurements and samples in the three lakes immediately downstream of Snap Lake, was to assess the spatial extent of the plume of treated effluent downstream of Snap Lake and to provide additional information for modelling of these downstream waterbodies. The purpose of the 2013 Nutrient Assessment was to evaluate laboratory accuracy of nutrient results and potential differences in nutrient results in Snap Lake due to different sampling methods. The 2013 Nutrient Assessment was a targeted follow-up study to a similar study completed in 2012. A limited number of nutrient spike samples (i.e., samples of known concentration) and samples from Snap Lake collected using both water quality and plankton sampling methods were sent to the three laboratories used to analyze nutrients in the AEMP.

3.2 Methods

3.2.1 Field Surveys for AEMP Sampling

3.2.1.1 Locations of Sampling Stations

The focus of the 2013 AEMP Design Plan, effective April 2013, shifted from evaluating spatial and seasonal trends in Snap Lake to monitoring trends over time and changes downstream of Snap Lake (De Beers 2014a). Therefore, the number of water quality stations monitored in Snap Lake during February and March 2013, when the 2005 AEMP Design Plan was still in effect, was reduced in April 2013 onwards, when the 2013 Design Plan was implemented (Figure 3-1). In February and March, 19 stations were monitored in Snap Lake, including 3 diffuser stations (SNP 02-20d, SNP 02-20e, and SNP 02-20f), 12 main basin stations (SNAP03, SNAP04, SNAP05, SNAP06, SNAP07, SNAP08,

SNAP09, SNAP10, SNAP11A, SNAP12, SNAP26, and SNAP28) and 4 northwest arm stations (SNAP02A, SNAP20B, SNAP23, and SNAP29). In April 2013, sampling in Snap Lake was discontinued at half of the main basin stations (SNAP04, SNAP07, SNAP10, SNAP12, SNAP26, and SNAP28) as per the approval of the 2013 AEMP Design Plan (MVLWB 2013b).

The three diffuser stations are located approximately 200 metres (m) away from the diffuser outlet, which discharges treated minewater into the main basin of Snap Lake. These three stations are intended to measure the concentration of the treated effluent plume after initial mixing.

The main basin, which was considered as one area for the purposes of reporting 2013 data, has historically been split into a near-field area, which included the three diffuser stations, a mid-field area, and a far-field area based on the historical gradient in the treated minewater plume observed in these areas. Since 2011, the gradient within the main basin has been diminishing and, in 2013, the gradient was undetectable, indicating that the main basin area beyond the diffuser stations can be treated as an area of approximately homogenous exposure to treated effluent.

The northwest arm is connected to the main basin of Snap Lake by a narrow area and has limited mixing in with the main basin. The water quality in the northwest arm has generally been the least influenced by treated effluent. This is likely because the area has limited hydraulic connectivity to the main basin of Snap Lake, due to the shallow depth at the narrows between the main basin and the northwest arm. The limited hydraulic connectivity is especially evident during winter when the northwest arm may be physically disconnected from the main body because of ice blockage over much of the narrows. However, water quality in the northwest arm has been increasingly influenced by treated effluent. Monitoring in the northwest arm is used to identify trends related to treated effluent exposure and potential seepage and overflow from the portion of the Mine site that is adjacent to the northwest arm, which includes the North Pile.

Field data were also collected at two additional stations (SNAP07 and SNAP15) in Snap Lake as part of the benthic invertebrate monitoring program (Section 6). The field methods used for collecting field water quality profiles at the benthic invertebrate stations, including monitoring frequency, are discussed in more detail in the benthic invertebrate field survey section (Section 6.2.1).

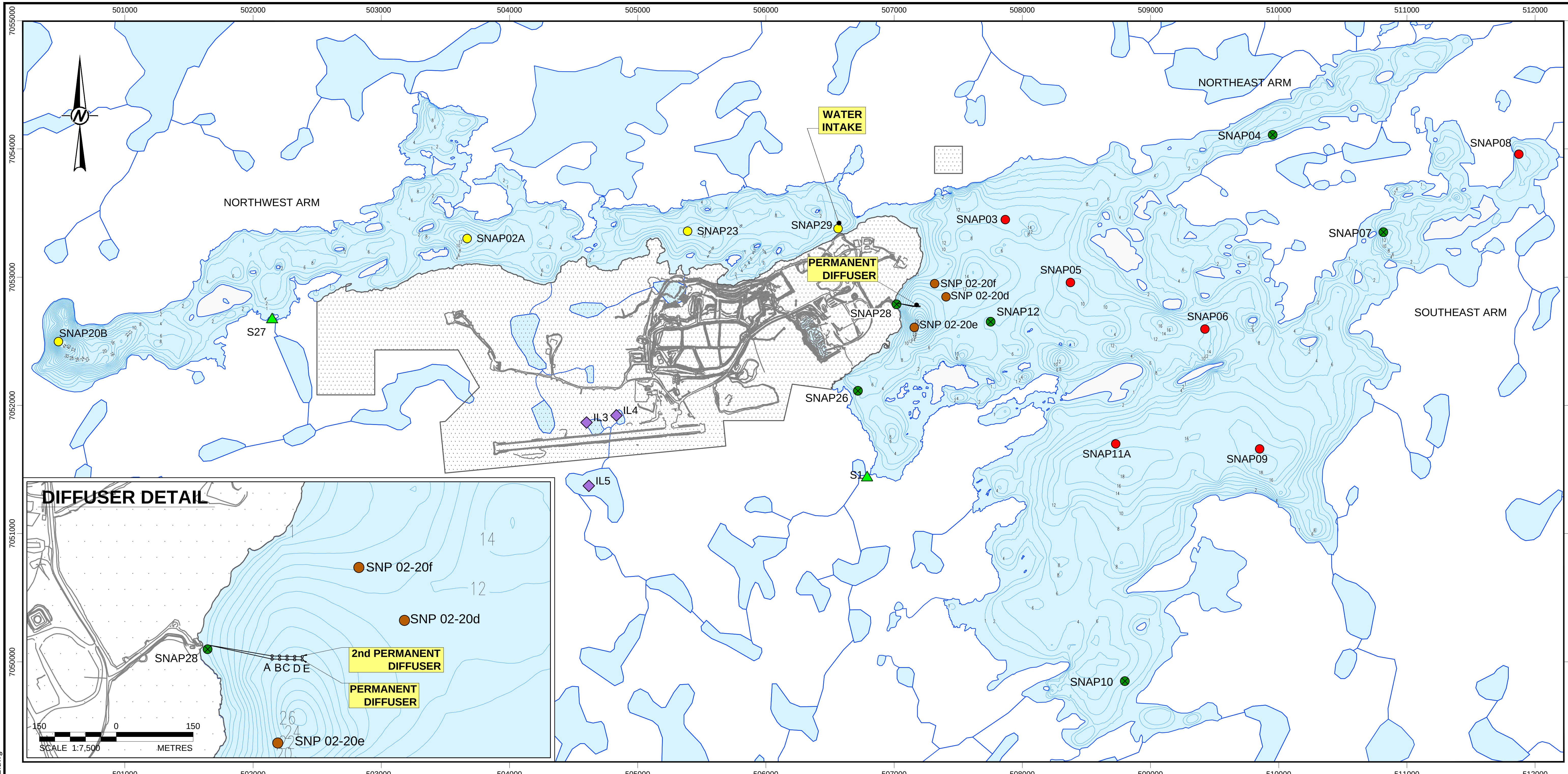
Eighteen stations located outside Snap Lake were also sampled as part of the AEMP in 2013:

- Station KING01 is located approximately 25 km downstream of Snap Lake in the Lockhart River system, upstream of King Lake (Section 1, Figure 1-2). Monitoring at KING01 is conducted to evaluate water quality at a location downstream of Snap Lake.
- Three inland lake (IL) stations (IL3, IL4, and IL5) are located towards the southwest end of the Mine property near the airstrip (Figure 3-1). These three stations are monitored to assess the potential for acidification in small waterbodies on the Mine property.

- Two watercourse stations, Streams S1 and S27, are located on major tributaries flowing into Snap Lake (Figure 3-1). These stations are monitored to provide an estimate of natural watershed loadings to Snap Lake and to assess the potential for acidification due to air emissions.
- Five water quality stations in the main basin of reference lake Northeast Lake (NEL01, NEL02, NEL03, NEL04, and NEL05) are monitored to identify local water quality changes that may not be influenced by Mine activities (Figure 3-2).
- Five water quality stations in the main basin of a second reference lake, Lake 13 (LK13-01, LK13-02, LK13-03, LK13-04, and LK13-05), are monitored to allow additional comparisons with reference lake water quality and to provide supporting information for other components (sediment, plankton, benthic invertebrates, and fish; Figure 3-3).
- Two reference dissolved oxygen (DO) profile stations in Northeast Lake (NEL06) and Lake 13 (LK13-06) are monitored to provide deep-water comparisons of DO concentrations between the two reference lakes and Snap Lake (Figures 3-2 and 3-3, respectively).

During the 2013 February sampling program, water quality field profiles were collected approximately 240 m away from the standard monitoring location for NEL06 in Northeast Lake. Water quality profile data involved measurements of the following field parameters: pH; specific conductivity, hereafter referred to as conductivity; DO; and, water temperature. Field profile data collected at the non-standard NEL06 station were similar to field profile data collected at the other locations in Northeast Lake in February; therefore, the data were considered valid and included in the assessment.

During the 2013 May and July sampling programs, water samples and field profiles were collected approximately 250 to 500 m away from the sampling locations described in the 2013 AEMP Design Plan (De Beers 2014a) for LK13-03, LK13-05, and LK13-06 in Lake 13. Water quality data collected at these non-standard locations in May and July were similar to water quality data collected at the other locations in Lake 13 in May and July, respectively; therefore, the data collected at the non-standard locations in Lake 13 were considered valid and included in the assessment.



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LEGEND

- Depth Contour (m)
- Watercourse
- Waterbody
- Snap Lake Mine Footprint
- Water Quality Monitoring Station : Diffuser
- Water Quality Monitoring Station : Main Basin
- Water Quality Monitoring Station Discontinued as of April 2013
- Water Quality Monitoring Station : Northwest Arm
- Inland Lake Station
- Tributary to Snap Lake Station

REFERENCES

Digital Map from Mackay lake, Northwest Territories, produced by Department of Energy, Mines and Resources. Map 75M, Original scale 1:250,000, projection : Transverse Mercator, Datum : NAD83, coordinate system : UTM zone 12.

Snap Lake outline and islands were corrected to LANDSAT 7 satellite image 45/15, dated September 2, 2000. provided by Geobase.

Bathymetry was created in Surfer 8 using sonar data from the 2002 North Lakes program (Golder) and 2005 transect data from the Reference Lake Search program (Golder).

SNAP LAKE MINE

Water Quality Monitoring Stations in and Adjacent to Snap Lake, 2013

PROJECTION: UTM Zone 12
DATUM: NAD83

500 0 500
SCALE 1:30,000 METRES

DE BEERS
GROUP OF COMPANIES

FILE No: 1313490001-7010_3-1
DATE: February 20, 2014
JOB No: 14-1349-0004
REVISION No: A
OFFICE: GOLDER VICTORIA
DRAWN: JEF
CHECK: PMC

Figure 3-1

LEGEND

- Depth Contour (m)
- Watercourse
- Waterbody
- Water Quality AEMP Monitoring Station
- Water Quality Profile Station

REFERENCES

Digital Map from Mackay lake, Northwest Territories, produced by Department of Energy, Mines and Resources. Map 75M, Original scale 1:250,000, projection : Transverse Mercator, Datum : NAD83, coordinate system : UTM zone 12.

Lake outline and islands were corrected to LANDSAT 7 satellite image 45/15, dated September 2, 2000. provided by Geobase.

Bathymetry was created in Surfer 8 using sonar data from the 2002 North Lakes program (Golder) and 2005 transect data from the Reference Lake Search program (Golder).

SNAP LAKE MINE

Water Quality Monitoring Stations in Northeast Lake, 2013

PROJECTION: UTM Zone 12	DATUM: NAD83		DE BEERS GROUP OF COMPANIES
SCALE 500 0 500 METRES			
FILE No: 1313490001-7010_3-2		DATE: March 25, 2014	
JOB No: 14-1349-0004	REVISION No: A		Figure 3-2
OFFICE: GOLDER VICTORIA	DRAWN: JEF	CHECK: PMC	

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LEGEND

- Depth Contour (m)
- Watercourse
- Waterbody
- Water Quality AEMP Monitoring Station
- Water Quality Profile Station

REFERENCES

Digital Map from Mackay lake, Northwest Territories, produced by Department of Energy, Mines and Resources. Map 75M, Original scale 1:250,000, projection : Transverse Mercator, Datum : NAD83, coordinate system : UTM zone 12.

Bathymetry was created in Surfer 8 using sonar data from the 2002 North Lakes program (Golder) and 2005 transect data from the Reference Lake Search program (Golder).

SNAP LAKE MINE

Water Quality Monitoring Stations in Lake 13, 2013

PROJECTION: UTM Zone 12	DATUM: NAD83	DE BEERS GROUP OF COMPANIES
SCALE 500 0 500 METRES		
FILE No: 1313490001-7010_3-3		DATE: March 25, 2014
JOB No: 14-1349-0004	REVISION No: A	Figure 3-3
OFFICE: GOLDER VICTORIA	DRAWN: JEF	CHECK: PMC

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3.2.1.2 Water Quality Monitoring for the Supplemental Studies

Downstream Lakes Special Study

Water quality in the first three lakes downstream of Snap Lake, Downstream Lakes 1 and 2 (abbreviated DSL1 and DSL2), and Lac Capot Blanc, was monitored as part of the 2013 Downstream Lakes Special Study. The Downstream Lakes Special Study was included in the 2013 AEMP Design Plan (De Beers 2014a) and was conducted May, July, August, and September to collect information on bathymetry, water quality, sediment, and chlorophyll *a*. Methods and results of the 2013 Downstream Lakes Special Study are presented in Section 11.3.

Nutrient Assessment

The 2013 Nutrient Assessment was completed as a follow-up study to the work completed in 2012, which assessed discrepancies between the nutrient data collected for the AEMP water quality and plankton programs from 2008 to 2011. The assessment included sending samples of known concentrations for laboratory analyses and sampling for nutrients at seven existing AEMP stations within Snap Lake using both the water quality and plankton sampling methods. Detailed methods and results of the 2013 Nutrient Assessment are presented in Appendix 3B.

3.2.1.3 Sampling Frequency

The 2013 AEMP report includes water quality monitoring data collected between November 1, 2012, and October 31, 2013. The reporting period was chosen to allow ice-covered and open-water seasons to be analyzed together.

In 2013, the ice-covered season was defined as November 2012 to June 2013. The open-water season was defined as July to October 2013. The 2013 seasonal delineations are consistent with ice-covered and open-water seasons delineated in previous AEMP reports (De Beers 2006, 2007a, 2008a, 2009, 2010, 2011, 2012b, 2013a). In 2013, the January quarterly field program was re-scheduled to February due to extremely cold temperatures that rendered sampling unsafe.

Since January 2007, surveys in June, October, November, and December have typically not been conducted due to unsafe ice conditions. This modification to the initial AEMP sampling design followed consultation with the MVLWB (De Beers 2007b), and was retained in the 2013 AEMP Design Plan (De Beers 2014a). The modification included contingencies for safe ice conditions, so that sampling would be conducted. This was the case in 2009 and 2013, when sampling programs were completed in June 2009 and October 2013 because of the unseasonably safe ice conditions.

In 2013, monitoring frequencies at stations were consistent with the 2005 Design Plan in February and March 2013. From April 2013 onward, stations were monitored at frequencies described in the 2013 AEMP Design Plan (De Beers 2014a).

The monitoring frequency for each program area is outlined in Table 3-1. Additional details on requirements for sampling frequency for specific parameters are provided in Section 3.2.1.7.

Table 3-1 AEMP Water Quality Monitoring Frequency, 2013

Area[a]	Sampling Stations	Frequency: 2005 AEMP Design Plan (February and March)	Frequency: 2013 AEMP Design Plan (April to October)	2013 Sampling Period
Main basin	SNP 02-20d, SNP 02-20e, and SNP 02-20f	monthly sampling and field profile as conditions allow[b]		February 10 March 17 and 19 April 21 May 7 July 9 August 11 September 8 October 1
Main basin	SNAP03, SNAP04[c], SNAP05, SNAP06, SNAP07[c], SNAP08[d], SNAP09, SNAP10[c], SNAP11A, SNAP12[c], SNAP26[c], and SNAP28[c]	quarterly sampling and field profiles, and monthly field profiles during the ice-covered season	four times per year sampling and field profiles (once near the end of the ice-covered season and monthly between July and September)	February[e] 8, 9, and 11 March[f] 16, 18, and 20 May 5, 9, and 10 July 7, 8, and 10 August 8 and 9 September 5 and 6
Northwest arm	SNAP02A, SNAP20B, SNAP23, and SNAP29			
Inland lakes	IL3, IL4, and IL5	monthly sampling and field profiles during the open-water season		July 12 August 10 September 8
Watercourses (major tributaries to Snap Lake)	S1 and S27	twice weekly sampling and field measurements during spring freshet		May 19, 23, 24 and 27 June 3
		monthly sampling and field measurements during open-water conditions		July 8 and 10 August 9 and 11 September 5 and 8
Downstream	KING01	quarterly sampling and field profiles, and monthly field profiles during the ice-covered season	annual sampling and field profiles	February[e] 14 May 7
Northeast Lake	NEL01, NEL02, NEL03, NEL04, NEL05, and NEL06[g]	quarterly water sampling and field profiles, and monthly field profiles during the open-water season	four times per year sampling and field profiles (once during the ice-covered season and three time during the open-water season)	February[e] 12 May 6 July 16 August 14 September 14 and 15

Table 3-1 AEMP Water Quality Monitoring Frequency, 2013

Area[a]	Sampling Stations	Frequency: 2005 AEMP Design Plan (February and March)	Frequency: 2013 AEMP Design Plan (April to October)	2013 Sampling Period
Lake 13	LK13-01, LK13-02, LK13-03, LK13-04, LK13-05, and LK13-06[g]	N/A	four times per year sampling and field profiles (once during the ice-covered season and three time during the open-water season)	May 8 July 15 August 12 and 13 September 12 and 14

a) Area was classified as part of the 2013 AEMP Design Plan (De Beers 2014a).

b) Monthly when ice conditions allow. Sampling did not occur during break-up (i.e., June) or freeze-up (November). The January sampling program was cancelled due to the extremely cold weather which rendered sampling unsafe.

c) Sampling was discontinued as of April 2013, in compliance with the 2013 AEMP Design Plan (De Beers 2014a).

d) SNAP08 is located at the Snap Lake outlet.

e) Quarterly water sampling and field profiles were completed as part of the 2005 AEMP Design Plan (De Beers 2005a).

f) Monthly field profiles were completed as part of the 2005 AEMP Design Plan (De Beers 2005a).

g) Northeast Lake station NEL06 and Lake 13 station LK13-06 were monitored for deep water dissolved oxygen comparison.

N/A = Lake 13 was not part of the 2005 AEMP Design Plan

AEMP=Aquatic Effects Monitoring Plan; SNP=surveillance network program; SNAP = Snap Lake; IL=inland lake; KING=King Lake; NEL=Northeast Lake; LK13=Lake 13.

3.2.1.4 Field Program Logistics

Snap Lake, Northeast Lake, and Lake 13 stations were accessed by snowmobile during ice-covered conditions and by boat during open-water conditions. A helicopter was used to sling the boat and transport the crews to Northeast Lake and Lake 13 during the open-water season. A helicopter was also required to access downstream station KING01. The inland lakes and Streams S1 and S27 were accessed by truck and on foot during open-water conditions. Streams S1 and S27 were accessed by snowmobile during spring freshet while ice conditions permitted safe access.

Station locations were identified using a hand-held Garmin global positioning system (GPS) and Universal Transverse Mercator (UTM) coordinates in conjunction with topographical maps showing station locations.

3.2.1.5 Sample Collection

Water was sampled according to standard water quality methods (Environment Canada 2012). These methods represent accepted procedures for collecting water samples, conducting field measurements, recording field notes, calibrating instruments, and QA/QC (De Beers 2008b).

Water from specific sampling depths at the station locations was collected using a Teflon Kemmerer sampler for all metals[2] and petroleum hydrocarbon samples and a polyvinyl chloride (PVC) Kemmerer sampler for all other samples.

Snap Lake, Northeast Lake and Lake 13 Stations

In February and March 2013, water samples were collected at the diffuser stations and AEMP stations in Snap Lake and Northeast Lake at depths specified in the 2005 AEMP Design Plan (De Beers 2005a). Three samples were collected at each of the diffuser stations (SNP 02-20d, SNP 02-20e, and SNP 02-20f) in February and March 2013:

- Near the surface of the water, at approximately 0.3 m below the surface during open-water sampling, or 0.3 m below the bottom of the ice layer during ice-covered sampling;
- At the depth of maximum conductivity[3], or at mid-depth in the water column if a vertical conductivity gradient was not observed; and,
- At 1.0 m above the lake bottom.

As of April 2013, one sample was collected at all stations in Snap Lake, Northeast Lake and Lake 13 (De Beers 2014a) from the depth of maximum conductivity, or at mid-depth if no conductivity gradient was

[2] The term “metals” includes metalloids (e.g., arsenic) and non-metals (e.g., selenium).

[3] Vertical conductivity gradients were not identified in Snap Lake and Northeast Lake in February 2013; therefore, mid-depth water samples were collected at each station as per the 2005 AEMP Design Plan (De Beers 2005a).

observed (i.e., conductivity values are consistent through the water column) (De Beers 2014a). A vertical conductivity gradient existed if the difference between highest value and lowest value was greater than 10 percent (%) of the highest value.

Streams, Inland Lake, and the Downstream Station

Surface water grab samples were collected at Streams S1 and S27, and the inland lake stations IL3, IL4, and IL5 during open-water conditions; samples were collected at approximately 0.3 m below the water surface. At the AEMP downstream station KING01, which was sampled during winter, grab samples were collected 0.3 m below the bottom of the ice.

Open-Water Sampling

During the open-water sampling season, water collected at each station was poured directly from the Kemmerer samplers into sampling bottles, with the exception of samples that required filtering. The samples that required filtering were dissolved metals, dissolved organic phosphorus, total dissolved phosphorus, and hexavalent chromium. Water that required filtering was poured from the Kemmerer sampler into clean 1 litre (L) laboratory grade sampling containers and filtered when the field crew returned to the De Beers water processing facility at the Mine at the end of the sampling day.

Total mercury samples were collected in 125 millilitre (mL) Teflon bottles and submitted to Flett Research Limited (Flett) (Section 3.2.1.7). Methyl mercury samples were collected in 250 mL glass or Teflon bottles. Flett supplied bottles filled with 0.4% hydrochloric acid solution; this solution was poured out and the bottles were rinsed three times with sample water before filling. Special instructions for mercury sampling procedures provided by Flett were followed for all samples for mercury analyses.

Ice-Covered Sampling

During the ice-covered sampling season, a gasoline-powered ice auger was used to drill a hole in the ice so that Kemmerer samplers could be lowered through the hole into the water column to collect water samples. During the ice-covered sampling, water from the Kemmerer samplers was poured into 4 L laboratory-grade sampling containers instead of individual sampling bottles. This modification reduced complications associated with attempting to fill several small bottles in temperatures well below freezing, and reduced the chances of contamination in the field. Individual sample bottles were then filled from the 4 L containers when the crew returned to the De Beers water processing facility at the end of the sampling day.

Toxicity Sampling

Four treated effluent samples were collected from the permanent WTP for quarterly toxicity testing in accordance with the Water Licence (MVLWB 2004, 2012) (i.e., January, May, September, and October 2013). The effluent samples were submitted to HydroQual Laboratories (HydroQual) in Calgary, Alberta (AB) and tested for acute and chronic toxicity (Section 3.2.1.7).

In 2013, toxicity samples were collected twice from Snap Lake at the three diffuser stations and submitted for chronic toxicity testing. To meet the requirements outlined in the Water Licence (MVLWB 2004, 2012, 2013a), sampling occurred once during ice-covered conditions (May), and once during open-water conditions (September). Samples were collected at the depth of maximum conductivity or at mid-depth, if no conductivity gradient was observed. Details on toxicity sample collection and supporting field data are provided in Appendix 3F.

3.2.1.6 Collection of Supporting Field Measurements

Field measurements of DO, pH, water temperature, and conductivity were collected using a YSI 650 Multi-parameter Display System (MDS) water quality meter with a YSI 600 Quick Sample (QS) multi-parameter water quality probe. A 30 m cable was used with the YSI meter for depth profiles. Field water quality profiles were collected every 0.5 m at stations with depths less than 5 m, and every 1 m at stations with depths greater than 5 m. Station number, UTM coordinates, date, time of collection, and weather were also recorded at each station. A summary of the field water quality profile measurements recorded for the AEMP is provided in Table 3-2.

Other field data collected were ice depth during ice-covered conditions and Secchi depth during open-water conditions. Ice depth was measured at each station using an ice-thickness gauge before sampling, and Secchi depths were measured using a 20 centimetre (cm) diameter Secchi disk, consistent with the method described in Dodds and Whiles (2010).

Water was collected in 300 mL glass bottles for Winkler titrations to confirm field measurements of DO.

Table 3-2 Summary of Field Parameters Monitored at Each AEMP Station

Category	Station	Parameter
Field profiles	lake stations (Snap Lake, Northeast Lake, and Lake 13)	water temperature, DO, pH, conductivity
Single (spot) measurements	lake stations (Snap Lake, Northeast Lake, and Lake 13)	total water depth, ice and snow depths during ice-covered conditions, Secchi depth during open-water conditions, wind and weather conditions during all sampling events
Single (spot) measurements	downstream station (KING01), streams S1 and S27, and Inland Lake stations (IL3, IL4, IL5)	water temperature, DO, pH, conductivity, wind and weather conditions

m = metre; DO = dissolved oxygen; KING = King Lake; IL = inland lake.

3.2.1.7 Laboratory Analyses

Water quality parameters, applicable sampling stations, and monitoring frequency of different parameter categories were adjusted in April 2013 as part of the 2013 AEMP Design Plan (De Beers 2014a). The water samples collected in February and March 2013 were collected at the locations, and analyzed for the parameters described in the 2005 AEMP Design Plan (De Beers 2005a); details are provided in the 2012 AEMP Report (De Beers 2013a). Water quality parameters, stations, and sampling frequency implemented as of April 2013, are summarized in Table 3-3.

The majority of water samples were submitted to ALS Canada Ltd. (ALS) in Edmonton, AB. Samples for ultra-low level mercury and methyl mercury analyses were submitted to Flett in Winnipeg, Manitoba (MB). Flett was selected for the ultra-low level mercury analyses because they could provide the low detection limits required for comparison to applicable guidelines and/or EAR predictions. Samples for *Escherichia coli* (*E. coli*) analysis were sent to Taiga Environmental Laboratory in Yellowknife, Northwest Territories (NWT), to meet required holding times. Maxxam Analytics in Burnaby, British Columbia (BC) was used for inter-laboratory comparisons of sample results. Toxicity samples were submitted to HydroQual in Calgary, AB. The parameter groups are defined in Table 3-3 and the analytical services provided by each laboratory in 2013 are:

- ALS in Edmonton: conventional and physical parameters, measured and calculated total dissolved solids (TDS) and major ions, standard and additional nutrients, ultra-low total and dissolved metals by collision cell inductively coupled plasma mass spectrometry (CCMS), total oil and grease by infrared analysis, and biochemical oxygen demand (BOD);
- Flett in Winnipeg: ultra-low level total mercury and methyl mercury, as per United States Environmental Protection Agency (USEPA) (2002) and USEPA (2001), respectively;
- Taiga Environmental Laboratory in Yellowknife: *E. coli*;
- Maxxam in Burnaby: conventional and physical parameters, measured and calculated TDS and major ions, standard and additional nutrients, ultra-low total and dissolved metals by CCMS, hexavalent chromium, organics, *E. coli*, and BOD; and,
- HydroQual in Calgary: chronic toxicity analyses were conducted on the diffuser station samples using a water flea species, *Ceriodaphnia dubia*, and an algae species, *Pseudokirchneriella subcapitata*; chronic toxicity, as above, and acute toxicity analyses were conducted on the final treated effluent. Acute toxicity tests were conducted with Rainbow Trout, *Oncorhynchus mykiss*, and a water flea, *Daphnia magna* (details provided in Appendix 3F).

Before shipping the samples to the relevant laboratories, a subset of the water samples required filtering and preserving. The subset involved samples collected for dissolved organic phosphorus, total dissolved phosphorus, dissolved metals, and hexavalent chromium analyses. These samples were filtered in the De Beers water processing facility using a Geopump2 filter unit, laboratory-grade silicon tubing, and 0.45 micrometre (µm) Waterra filters, which are certified high capacity in-line groundwater sampling capsules. Preservatives, supplied by the laboratory to which the samples were being sent, were added to samples as required, following standard protocols for specific parameters (APHA 2012).

Table 3-3 Summary of the 2013 AEMP Design Plan Water Quality Parameters, Stations, and Sampling Frequency

Parameter Categories	Parameter	Snap Lake – Diffuser Stations	Snap Lake – Main Basin and Northwest Arm	Reference Lakes (Northeast Lake and Lake 13)	Inland Lake Stations	Tributary Stations	Downstream Station
		SNP02-20d; SNP02-20e; SNP02-20f	Main Basin: SNAP03; SNAP05; SNAP06; SNAP08; SNAP09; SNAP11A	NEL01; NEL02; NEL03; NEL04; NEL05; NEL06[a]	IL3; IL4; IL5	S1; S27	KING01
			Northwest Arm: SNAP02A; SNAP23; SNAP20B; SNAP29	LK13-01, LK13-02, LK13-03, LK13-04, LK13-05, LK13-06[a]			
Field Measurements/ Profiles	Field pH, specific conductivity, dissolved oxygen, and temperature	monthly (at 1-m intervals from surface to bottom)[b]	April/May, July, August, September (at 1-m intervals from surface to bottom)	April/May, July, August, September (at 1-m intervals from surface to bottom)	monthly during open-water conditions (surface)	twice weekly during spring freshet and monthly during open-water conditions	April/May
Physical and conventional parameters, TDS and major ions	total suspended solids; pH; turbidity; conductivity, TDS (calculated and measured); calcium; magnesium; sodium; chloride; sulphate; bicarbonate; carbonate; fluoride; potassium; hydroxide; reactive silica (as SiO_2); hardness; alkalinity; acidity; ion balance	monthly[b]	April/May, July, August, September	April/May, July, August, September	monthly during open-water conditions	twice weekly during spring freshet and monthly during open-water conditions	April/May
Nutrients	total and dissolved phosphorus; total organic carbon; ortho-phosphate as P; total and dissolved organic phosphorus; total and dissolved inorganic phosphorus; total ammonia (as nitrogen [N]); nitrate (as N); nitrite (as N); nitrate/nitrite (as N); total Kjeldahl nitrogen (as N)	monthly[b]	April/May, July, August, September	April/May, July, August, September	monthly during open-water conditions for nitrogen nutrients[c]	weekly during spring freshet and monthly during open-water conditions for nitrogen nutrients[c]	April/May
Metals	total and dissolved metals (Al; Sb; As; Ba; Be; Bi; B; Cd; Cs; Cr; Co; Cu; Fe; Pb; Li; Mn; Hg; Mo; Ni; Se; Ag; Sr; Tl; Ti; U; V; Zn) and hexavalent Cr	monthly[b]	April/May, September[d]	April/May, September[d]	not applicable	weekly during spring freshet and monthly during open-water conditions[d]	April/May[d]
Other parameters	methyl mercury and biochemical oxygen demand (BOD)	monthly[b]	not applicable; except BOD at SNAP08	not applicable	not applicable	not applicable	not applicable
Organics	BTEX (benzene; toluene; ethylene; xylene); total oil and grease; total extractable hydrocarbons; total volatile hydrocarbons F1 (without BTEX) and F2 (without BTEX	monthly[b]	not applicable	not applicable	not applicable	not applicable	not applicable
Biological	*Escherichia coli*	monthly[b]	not applicable	not applicable	not applicable	not applicable	not applicable
	Microcystin-LR	not applicable	January, April, July, August, September at SNAP29[e] only	not applicable	not applicable	not applicable	not applicable
Toxicity	*Ceriodaphnia dubia; Pseudokirchneriella subcapitata*	twice per year (April/May, September)	not applicable	not applicable	not applicable	not applicable	not applicable

Note:

a) Field measurements/profiles only.

b) Monthly when ice conditions allow. Sampling did not occur during break-up (i.e., June) and freeze-up (i.e., November) during the 2013 AEMP sampling period.

c) Nitrogen nutrients = total ammonia (as nitrogen [N]); nitrate (as N); nitrite (as N); nitrate/nitrite (as N); total Kjeldahl nitrogen (as N).

d) Samples were analyzed for total metals; dissolved metals samples were archived and only analyzed if a total metal was above an AEMP benchmark.

e) SNAP29 = water intake location.

AEMP = Aquatic Effects Monitoring Program; April/May = monitoring will occur between April 1 and May 31 with the intention of monitoring water quality near the end of ice-covered conditions. In 2013, samples from the late ice-covered period were collected in May; SNP = surveillance network program; IL= inland lake; SNAP= Snap Lake; NEL= northeast lake; LK13 = Lake 13; KING = King Lake; TDS = total dissolved solids; SiO_2 = silicate; P = phosphorus; N = nitrogen; BOD = biochemical oxygen demand; m = metre; SNP= Surveillance Network Program; BTEX = benzene, toluene, ethylbenzene, xylene; Al = aluminum; Sb = antimony; As = arsenic; B = boron; Ba = barium; Be = beryllium; Bi = bismuth; Cd = cadmium; Cr = chromium; Co = cobalt; Cs = cesium; Cu = copper; Fe = iron; Pb = lead; Li = lithium; Mn = manganese; Hg = mercury; Mo = molybdenum; Ni =nickel; Se = selenium; Ag = silver; Sr = strontium; Tl = thallium; Ti = titanium; U = uranium; V = vanadium; Zn = zinc.

3.2.2 Data Analyses

3.2.2.1 Approach

Analyses of the 2013 water quality data focused on answering six key questions (Table 3-4):

- Are concentrations or loads of key water quality parameters in discharges to Snap Lake consistent with EAR predictions and below Water Licence limits?
- Are concentrations of key water quality parameters in Snap Lake below AEMP benchmarks and Water Licence limits?
- Which water quality parameters are increasing over time in Snap and nearby waterbodies, and how do concentrations of these parameters compare to AEMP benchmarks, concentrations in reference lakes, EAR predictions, and subsequent modelling predictions?
- Are spatial and seasonal patterns in water quality in Snap Lake and downstream waterbodies consistent with predictions presented in the EAR and subsequent modelling predictions?
- Is there evidence of acidification effects from the Mine on nearby waterbodies?
- Is water from Snap Lake safe to drink?

The methods used to answer the six key questions are summarized in Table 3-4 and described in more detail in Sections 3.2.2.2 to 3.2.2.7.

Table 3-4 Overview of Analysis Approach for Water Quality Key Questions

Key Question	Overview of Analysis Approach
1. Are concentrations or loads of key water quality parameters in discharges to Snap Lake consistent with EAR predictions and below Water Licence limits?	Treated effluent discharge to Snap Lake was compared to EAR predictions and Water Licence limits. Temporal trends in treated effluent concentrations and loads were investigated. Toxicity of the treated effluent was evaluated. Other inputs (e.g., seepage, runoff, spills) were discussed, where appropriate.
2. Are concentrations of key water quality parameters in Snap Lake below AEMP benchmarks and Water Licence limits?	Concentrations of water quality parameters (i.e., maximums, whole-lake averages) were compared to AEMP benchmarks and the TDS Water Licence limit. Instances where concentrations were above, or below for pH and DO, AEMP benchmarks or limits were identified and qualitatively assessed for potential Mine-related causes. Results of toxicity testing of water from the mixing zone were also reviewed for chronic toxicity.
3. Which water quality parameters are increasing over time in Snap Lake and nearby waterbodies, and how do concentrations of these parameters compare to AEMP benchmarks, concentrations in reference lakes, EAR predictions, and subsequent modelling predictions?	An analysis of temporal patterns in water quality was completed for DO, TP, parameters that were higher in Snap Lake relative to the reference lakes, and parameters that are significantly correlated with conductivity in Snap Lake. Comparisons were made to the normal range observed prior to treated effluent discharge as well as reference lake concentrations. The parameters above AEMP benchmarks were assessed for apparent increasing trends (or decreasing trends as for DO and pH) in Snap Lake, including a comparison to EAR predictions and updated model results. A statistical test (e.g., Seasonal Kendall or other appropriate test) was used when the presence or absence of a trend was uncertain and additional confirmation was required.

Table 3-4 Overview of Analysis Approach for Water Quality Key Questions

Key Question	Overview of Analysis Approach
4. Are spatial and seasonal patterns in water quality in Snap Lake and downstream waterbodies consistent with predictions presented in the EAR and subsequent modelling predictions?	Qualitative assessments of horizontal, vertical, and seasonal patterns in Snap Lake water quality were completed for field parameters, TDS, major ions, nutrients, and metals. Where patterns existed, the potential for Mine-related causes was qualitatively assessed. An assessment of the data collected downstream of Snap Lake was completed to delineate the extent of the treated effluent plume as part of the Downstream Lakes Special Study. Conductivity was used as a tracer of treated effluent exposure. An analysis of temporal patterns in conductivity and TDS at KING01 (the most downstream AEMP station) was completed.
5. Is there evidence of acidification effects from the Mine on nearby waterbodies?	Water quality data from inland lake stations IL3, IL4, and IL5, streams S1 and S27 were reviewed to identify any changes in stream water quality related to mining activities, including potential acidification effects, and to document loadings to Snap Lake from this source.
6. Is water from Snap Lake safe to drink?	Water quality data from Snap Lake and station SNP 02-15 (the water intake) were compared to Canadian health-based drinking water guidelines.

EAR = Environmental Assessment Report; TDS = total dissolved solids; DO = dissolved oxygen; TP = total phosphorus; KING = King Lake; IL = inland lake; AEMP = Aquatic Effects Monitoring Program; SNP = Surveillance Network Program.

The 2013 AEMP Design Plan describes additional assessments for compiling information from these six key questions to inform the WOE and Action Level assessments, as part of the AEMP Response Framework. The WOE assessment, which integrates the findings from all the AEMP disciplines to determine any effects are occurring in the lake due to the Mine and their significance, is described in Section 12. The methods for the water quality Action Level assessment, which provides a systematic approach to responding to the water quality results in the AEMP, are provided Section 3.2.2.8.

The results and conclusions from the 2013 AEMP, organized by key question, and the Action Level assessment are provided in Sections 3.4 and 3.5, respectively.

3.2.2.2 Key Question 1: Are Concentrations or Loads of Key Water Quality Parameters in Discharges to Snap Lake Consistent with EAR Predictions and below Water Licence Limits?

Calculations of Treated Effluent Concentrations and Loadings

Temporal plots of parameter concentrations and loadings (from both the WTP and the temporary water treatment plant [TWTP], as applicable) were prepared. Comparisons of discharge quality to Water Licence limits and EAR predictions, determination of dilution factors, and a summary of the toxicity test results are provided.

The discharge volume and water quality data used to answer Key Question 1 in the 2013 AEMP were representative of combined treated effluent from the WTP, which included treated discharges from the TWTP (minewater) and sewage treatment plant (treated domestic waste water). The treated effluent from the TWTP was redirected through the WTP in March 2012. During the 2013 AEMP reporting period, all treated effluent was discharged from the WTP to Snap Lake through one or two diffusers

(Section 2.2.1.4). Effluent results collected between January 1, 2013 to December 31, 2013 at SNP 02-17B are presented herein to align with the SNP annual report reporting period.

Comparisons to Water Licence Limits

Parameters that have Water Licence limits are total suspended solids (TSS), three nitrogen compounds (ammonia, nitrate, and nitrite), two major ions (chloride and sulphate), six metals (aluminum, chromium, copper, lead, nickel, and zinc), and a metalloid (arsenic) (MV2011L2-0004: MVLWB 2013a) (Table 3-5).

The following additional limits apply at the final point of discharge (i.e., end-of-pipe):

- the pH level is to be maintained within the range of 6 to 9 pH units;
- the monthly average limit for extractable petroleum hydrocarbons is 4.6 milligrams per litre (mg/L) for F1 fractions (C_6-C_{10}) and 2.1 mg/L for F2 fractions (C_{11}-C_{16}); and,
- total phosphorus (TP), ammonia and nitrate annual loadings in kilograms per year (kg/y) (Table 3-5).

Table 3-5 Water Licence Limits for Treated Effluent

Parameter	Maximum Concentration of Any Grab Sample (mg/L)	Average Monthly Limit (mg/L)	Average Annual Loading (kg/y)
Total Suspended Solids	14	7	-
Ammonia, as N	20	10	187,000
Total Phosphorus, as P	-	-	256
Nitrite, as N	1	0.5	-
Nitrate, as N	44	22	219,000
Chloride	620	310	-
Sulphate	150	75	-
Aluminum	0.2	0.1	-
Arsenic	0.014	0.007	-
Chromium	0.02	0.01	-
Copper	0.006	0.003	-
Lead	0.01	0.005	-
Nickel	0.1	0.05	-
Zinc	0.02	0.01	-

Source: Water Licence: MV2011L2-0004 (MVLWB 2013a)
- = limit not specified; mg/L = milligrams per litre; kg/y = kilograms per year.

For the parameters with Water Licence limits, both a “maximum concentration in any grab sample” and an “average monthly limit” are specified. A rolling average based on six samples, whereby each sample result and the previous five sample results were averaged, was compared to the average monthly limit (MVLWB 2013c). A rolling average was calculated and compared to the average monthly limit for parameters measured every six days (i.e., physical parameters, major ions, nutrients). For total metals, a monthly average, where all samples collected in one calendar month were averaged, was calculated until May 2012 and compared to the average monthly limit. Sampling frequency increased for metals with Water Licence limits in June 2012 from monthly to every six days, in accordance with Water Licence MV2011L2-0004 (MVLWB 2013a); a rolling average replaced the monthly average for comparison to the average monthly limit. For extractable petroleum hydrocarbons, which were sampled approximately monthly, a monthly value, or average when applicable, was compared to the average monthly limit.

All 2013 treated effluent data were plotted with the historical data collected at TWTP (SNP 02-17) and WTP (SNP 02-17B) from 2004 and from 2007, respectively. For the parameters with Water Licence limits, the rolling averages or monthly averages, or both, were also plotted for the WTP so that direct visual comparisons to Water Licence limits could be made.

Daily discharge volumes and loadings rates (kilograms per day) were calculated and reviewed for trends over time. Annual loading rates to Snap Lake from the WTP and TWTP treated wastewater discharge for TP, ammonia and nitrate were derived from the Water Licence Annual Report (De Beers 2014c).

Comparisons to Environmental Assessment Report Predictions and 2013 Modelling Predictions

Measured concentrations in the treated effluent in 2013 were compared to predicted EAR concentrations for treated effluent (De Beers 2002), as well as the updated effluent predictions generated as part of the 2013 Water Licence Amendment Application (De Beers 2013b).

In the 2013 water quality modelling update, a range of treated effluent concentrations were predicted between 2012 and 2028 from the permanent WTP (i.e., SNP 02-17B) using the Snap Lake site model (De Beers 2013b). The model scenarios were based on the expected range of groundwater inflow rates to the Mine and the expected range of TDS concentrations in the inflows to the Mine (Itasca 2013; De Beers 2013b):

- Lower Bound Scenario A: Based on minewater flows from the Base Case of the groundwater model – decreased flows from lake and connate water, using the arithmetic mean connate water TDS;
- Lower Bound Scenario B: Based on minewater flows from the Base Case of the groundwater model – decreased flows from lake and connate water, using the geometric mean connate water TDS;
- Upper Bound Scenario A: Based on minewater flows from Scenario 4 of the groundwater model – elevated lake and connate water flows using the arithmetic mean connate water TDS concentration; and,
- Upper Bound Scenario B: Based on minewater flows from Scenario 4 of the groundwater model – elevated lake and connate water flows using the geometric mean connate water TDS concentration.

Average annual concentrations were calculated for each modelling scenario using the predicted daily concentrations in 2013. Flow-weighted average concentrations were calculated based on measured data from SNP 02-17B and compared against the 2013 average concentrations from the four modelling scenarios. Flow-weighted average concentrations (Equation 3-1) were used as they are reflective of average conditions, rather than instantaneous concentrations.

Flow-weighted average concentrations that exceeded EAR predictions or the range in predicted concentrations from the modelling update were identified. Loadings were calculated for parameters with mass-based units; parameters such as pH and conductivity were excluded. The combined weighted average used for comparison was calculated using Equation 3-1:

$$FWC_{WTP} = \Sigma(C_{WTPi} \times F_{WTPi}) / \Sigma(F_{WTPi}) \qquad \text{[Equation 3-1]}$$

where:

FWC_{WTP} = flow-weighted average concentration in the treated effluent from SNP 02-17B;

C_{WTPi} = concentration in the treated effluent from SNP 02-17B during sampling event i;

F_{WTPi} = daily discharge volume at SNP 02-17B associated with sampling event i; and,

i = sampling event.

Biological data for the treated effluent samples, including bacterial counts of *E. coli*, are presented as geometric means. Bacteria reproduce at an exponential rate in domestic waste water. Therefore, it is common to have an exceptionally wide range in bacterial coliform counts in some domestic waste water samples, such as 10 colony forming units per 100 millilitres (CFU/100 mL) to 100,000 CFU/100 mL. Compared to an arithmetic mean, the geometric mean is less sensitive to the effects of extreme values. Geometric means were calculated using Equation 3-2:

$$GM\ \bar{y} = (y1 \times y2 \times y3 \ldots yn)1/n \qquad \text{[Equation 3-2]}$$

where:

y = bacterial counts;

n = number of samples; and,

$GM\ \bar{y}$ = geometric mean.

Toxicity of Treated Effluent

Results of treated effluent toxicity tests for 2013 were included in this annual AEMP report and reviewed for trends and/or concentration response relationships (i.e., potential adverse effects increasing at higher concentrations of treated effluent). Adverse effects are considered to occur if there is more than a 25% (for a chronic test) or 50% (for a chronic or acute test) decrease in mean response in an undiluted sample, depending on the endpoint.

Dilution Factors

Multiple diffusers were used to discharge treated effluent to Snap Lake in 2013. The diffusers operating in 2013 were intended to maximize the potential for initial mixing of the treated effluent discharged to Snap Lake, thereby reducing peak concentrations of the treated effluent near the diffuser. Operating multiple diffusers may influence the dilution of the treated effluent discharged to Snap Lake because the different diffusers may provide different mixing characteristics. However, the diffusers do not influence total loadings to Snap Lake or average concentrations in the lake. The dilution from the operating diffusers was estimated quarterly by calculating dilution factors in February, May, July, August, and September. Dilution factors were calculated using TDS concentrations in the WTP discharge and from the annual monitoring program in Snap Lake, using Equation 3-3:

$$DF = (C_e - C_b)/(C_d - C_b) \quad \text{[Equation 3-3]}$$

where:

DF = minimum dilution factor of the diffuser(s);

C_e = combined flow-weighted average TDS concentration in the treated effluent from SNP 02-17B;

C_d = maximum TDS concentration at the three diffuser stations SNP 02-20d, SNP 02 20e, and SNP 02-20f; and,

C_b = background lake concentration, represented by the average TDS concentrations from main basin stations[4]: SNAP03, SNAP05, and SNAP06 in Snap Lake.

The calculated dilution factors were then compared with predicted dilution factors in the EAR (De Beers 2002).

Other Inputs to Snap Lake

Inputs other than treated effluent (e.g., uncontrolled runoff, seepage, overland spills) can also negatively affect water quality in Snap Lake, although to a much lesser extent than the treated effluent discharge. The term “uncontrolled runoff” refers to water that collects in bogs and catchments, and may enter Snap Lake; these runoff areas are monitored as part of the SNP. The quality and quantity of uncontrolled runoff and groundwater are discussed in the 2013 Acid/Alkaline Rock Drainage (ARD) Appendix of the Water Licence Annual Report submitted to MVLWB (De Beers 2014c).

Four stations to be included in the calculation will be determined based on the observed spatial gradient for that year. If there is less than a 10% difference between concentrations at the diffuser station and SNAP08, all stations in the main basin will be included. As concentrations in the lake increase, less spatial gradient in the main basin is expected, and more stations in the main basin will be included in the calculation.

3.2.2.3 Key Question 2: Are Concentrations of Key Water Quality Parameters in Snap Lake below AEMP Benchmarks and Water Licence Limits?

AEMP Benchmarks

In the EAR, parameter concentrations in Snap Lake were predicted to remain below the aquatic life (e.g., CCME 1999) or site-specific benchmarks developed in the EAR, such as those specifically developed for three metals: copper, cadmium, and hexavalent chromium.

Since the time the EAR was prepared, three new Canadian Council of Ministers of the Environment (CCME) water quality guidelines (WQGs) for the protection of aquatic life have been developed (i.e., fluoride, chloride, and nitrate). These new WQGs have been incorporated into the AEMP water quality data comparisons. Water quality data collected in Snap Lake during 2013 were compared against “AEMP benchmarks”, which refers to a collective list of generic WQGs (i.e., CCME 1999) and EAR benchmarks (De Beers 2002). The list will continue to evolve as new WQGs are published or revised by the CCME and new information becomes available. Site-specific benchmarks developed for Snap Lake (e.g., TDS, chloride, fluoride, nitrate) as part of the AEMP Response Framework will be highlighted separately.

Maximum concentrations from 2013 were compared to the AEMP benchmarks, with the exception of pH, DO, and TP. The range of the observed pH values was compared to the AEMP benchmark range. Minimum DO concentrations were compared to the aquatic life WQG (CCME 1999). Dissolved oxygen concentrations in late winter in Snap Lake were also compared to values from the same period at reference stations in Northeast Lake and Lake 13. The range in whole-lake averages for TP collected using the water quality and plankton component methods was compared to the AEMP nutrient benchmark for phosphorus as per the 2013 AEMP Design Plan (De Beers 2014a). Using whole-lake averages for nutrient comparisons is appropriate when considering overall lake trophic status.

If results were above AEMP benchmarks, an attempt was made to determine the relevance of the elevated results to aquatic biota. Where appropriate, this involved additional comparison of whole-lake average concentrations to WQGs, or comparison to recommended site-specific water quality objectives (SSWQOs), with consideration of the information on which they were developed.

To provide the required information to assess whether a water quality Action Level has been triggered (see Section 3.2.2.8), two additional comparisons to AEMP benchmarks were completed:

- monthly averages of concentrations at the diffuser stations (SNP 02-20d, SNP 02-20e, SNP 02-20f) were compared to 75% of the AEMP benchmark; and,
- whole-lake average concentrations of total phosphorus were compared to 75% of the nutrient AEMP benchmark.

Whole-Lake Average Concentrations of Total Dissolved Solids

The EAR for the Mine predicted that water discharged to Snap Lake would increase concentrations of TDS and some major ions, nutrients, and metals in Snap Lake (De Beers 2002). The Water Licence requires that a whole-lake average TDS concentration be calculated quarterly, including data collected at Snap Lake monitoring stations, excluding the northwest arm stations, then compared with the compliance limit of 350 mg/L (MVLWB 2013a). In 2013, since all TDS concentrations were less than 350 mg/L, a mean of the depth-averaged means at all stations was used to calculate the whole-lake average. In future years, if the depth-averaged concentration at any station is above 350 mg/L, and a spatial pattern in TDS concentrations is apparent, then the calculation of whole-lake averages will also account for spatial patterns following the procedure outlined in the Water Licence (MVLWB 2013a).

Total dissolved solids concentrations can be measured directly by evaporating a known volume of filtered water and measuring the mass of the residue left after evaporation (APHA 2012; Method 2540). Alternatively, TDS concentration can be calculated from the summation of major ions in the sample (APHA 2012; Method 1030). As part of the AEMP, TDS will be included as both measured and calculated values, but only calculated TDS will be used in the assessment (De Beers 2014a, MVLWB 2013a).

Toxicity Testing for Snap Lake

The EAR predicted that no persistent chronic toxicity would occur in Snap Lake. Results for the sublethal endpoints from the chronic toxicity tests, *Ceriodaphnia dubia* reproduction, and *Pseudokirchneriella subcapitata* algal growth, were plotted and reviewed for trends. When possible, toxicity results were compared to water quality data from treated effluent and diffuser stations sampled on the same day. Additional details regarding toxicity testing and data analysis are provided in Appendix 3F.

3.2.2.4 Key Question 3: Which water quality parameters are increasing over time in Snap Lake and nearby waterbodies, and how do concentrations of these parameters compare to AEMP benchmarks, concentrations in reference lakes, EAR predictions, and subsequent modelling predictions?

The purpose of Key Question 3 is to provide context for and evaluate the relevance of increasing trends in water quality parameters in Snap Lake. The following methods were used to answer Key Question 3:

- identifying which parameters are increasing in Snap Lake by:
  - screening for parameters that are positively (or negatively for pH) correlated with conductivity and then visually evaluating temporal plots for these parameters at selected stations in Snap Lake and the reference lakes;
  - visually evaluating temporal plots at selected stations in Snap Lake and reference lakes for parameters that may not be correlated with conductivity but could still be increasing in Snap Lake

due to the Mine (e.g., TP and parameters that are higher in Snap Lake relative to the reference lake).

- comparing Snap Lake concentrations with predictions from both the EAR and the 2013 modelling update:
  - screening for key parameters that are above AEMP benchmarks;
  - visually comparing temporal plots for parameters above AEMP benchmarks at selected stations and on a whole-lake average basis to EAR predictions and 2013 modelling update predictions.
- using a statistical test to confirm the absence or presence of increasing trends for selected parameters at selected stations; and,
- reviewing vertical profiles of DO concentrations from different areas in Snap Lake over time.

Screening Based on Visual Evaluation of Temporal Plots

The EAR predicted that discharges of treated effluent from the Mine to Snap Lake would result in increases in concentrations of major ions, nutrients, and some metals throughout the lake, and slight decreases in DO in deep waters of Snap Lake. Increases in several parameters in Snap Lake have been demonstrated in previous AEMP reports (De Beers 2006, 2007a, 2008a, 2009, 2010, 2011, 2012b, 2013a).

To confirm apparent trends and identify other water quality parameters that may be increasing in Snap Lake due to the treated effluent, Pearson correlation coefficients were calculated between each parameter and conductivity using SYSTAT 13.00.05 (SYSTAT 2009) for AEMP data collected from 2004 to 2013. Conductivity was selected as an indicator of exposure to the treated effluent because:

- conductivity is a parameter that can easily and reliably be measured in the field and laboratory;
- conductivity has increased throughout Snap Lake from 2004 to 2013, directly related to the input of treated effluent; and,
- conductivity was used to evaluate the degree of treated effluent exposure for other monitoring programs, including sediment quality (Section 4) and benthic invertebrates (Section 6).

The Pearson correlation test was used to determine whether changes in laboratory conductivity in Snap Lake correspond to linear changes in the concentration of other monitored parameters. A *P*-value of 0.001 was used to identify those parameters that were significantly correlated with conductivity to account for the large number of correlations (104) and the large sample size (generally greater than 1,000 samples). In cases where data outliers, which were visually identified in the parameter dataset by plotting the parameter dataset against the conductivity dataset, appeared to be influencing the parameter correlation with conductivity, the outliers were removed, and the Pearson correlation test was re-run to determine whether they had an influence on the strength of the correlation. Parameters with 99% of values below detection limits in Snap Lake, which included four metals (beryllium, bismuth, cesium, and silver) and most organics, were not tested for trends (Appendix 3G; Table 3G-2). All parameters that significantly correlated with conductivity, based on the inclusion or exclusion of the outliers, were reviewed for temporal trends in Snap Lake. The strength of the correlations was classified as low ($r < 0.3$), moderate (r between 0.4 and 0.7), or high ($r > 0.7$) based on ranges provided by Hinkle et al. (2003). Parameters with moderate and high correlations with conductivity were considered chemical signatures of treated effluent exposure.

Temporal plots of concentrations of those parameters that were significantly positively correlated with conductivity were completed (Appendix 3G) for one or more representative stations in each area of Snap Lake and the reference lakes:

- SNAP13 and SNP 02-20e (located near the permanent diffuser, at the edge of the mixing zone);
- SNAP05, SNAP09, and SNAP08 (located in the main basin of Snap Lake);
- SNAP02 and SNAP02A (located in the northwest arm of Snap Lake); and,
- all stations in the reference lakes (Northeast Lake and Lake 13).

Stations SNAP13 (diffuser) and SNAP02 (northwest arm) were established in 2004 and monitored until 2006. These stations were then discontinued, moved slightly, and renamed SNP 02-20e and SNAP02A, respectively.

Data from both the historical and new stations were included to provide a longer dataset for the analyses. Total nitrogen (TN) and TP concentrations from the water quality and plankton components were not combined because they were collected using different methods. Total nitrogen and TP samples for the water quality component were collected near the lake surface, at mid-depth, or near the bottom of the lake and submitted to ALS for analysis; TN and TP samples for the plankton component were based on a depth-integrated sample within the euphotic zone (i.e. the top 6 m of the water column) and submitted to the University of Alberta Biogeochemical Analytical Service Laboratory (UofA) for analysis.

Each plot was visually examined to identify increasing trends by lake area. Parameters that correlated with conductivity and demonstrated an increasing trend within one or more lake areas were identified. Plots of pH were reviewed for both potential decreasing and increasing trends.

Temporal plots of nutrients (e.g., TP) and parameters that were higher in Snap Lake relative to the reference lake (See Key Question 4, Section 3.2.2.5) at each of the above stations were also reviewed, regardless of the strength of correlations with conductivity. Nutrient trends were reviewed because nutrients could increase in Snap Lake without showing a strong correlation with conductivity due to seasonal fluctuations in biological uptake and release of nutrients. Parameters that were higher in Snap Lake relative to reference lakes, and not correlated to conductivity, were also reviewed for temporal trends in case the parameter increased in Snap Lake but did not follow the typical pattern for effluent related parameters in Snap Lake.

Water quality data for Northeast Lake and Lake 13 were also visually reviewed for temporal trends. Notable changes in water quality are not expected in Northeast Lake or Lake 13; therefore, any changes over time in Snap Lake that do not occur in the reference lakes are likely related to the Mine. Changes that occur in all three lakes would be attributed to non-Mine-related regional effects, such as climate change or hydrological variation.

To provide additional context for assessing increasing trends and the required information to assess water quality Action Levels (see Section 3.2.2.8), Snap Lake water quality concentrations in 2013 that were outside the baseline normal range for Snap Lake (i.e., average between 1999 and 2004, plus or minus [±] two standard deviations [SD]) or outside the reference normal range (i.e., average from 2013 monitoring time period, ± two SD) for Lake 13 and Northeast Lake were also identified.

Comparison to Environmental Assessment Report Predictions and 2013 Water Licence Amendment Application Predictions

In the 2013 water quality modelling update completed as part of the 2013 Water Licence Amendment Application, concentrations for key parameters were predicted in Snap Lake (De Beers 2013c) for the four modelling scenarios outlined in Section 3.2.2.2:

- Lower Bound Scenario A;
- Lower Bound Scenario B;
- Upper Bound Scenario A; and,
- Upper Bound Scenario B.

Whole-lake average observed concentrations of water quality parameters in Snap Lake in 2013 were compared to predictions from the EAR and 2013 water quality modelling update. Temporal trends for parameters that were above AEMP benchmarks in 2013 were compared to available trends predicted in the EAR and the 2013 modelling update. Temporal plots of observed data were superimposed on the EAR and updated 2013 modelling prediction plots, and were reviewed to determine whether values were increasing as expected. The 2013 water quality modelling update provided predictions from multiple scenarios (De Beers 2013c), whereas the EAR provided water quality predictions from one Mine scenario near the diffuser, in the main basin and at the outlet of Snap Lake. Therefore, the minimum and maximum predicted concentrations from all four scenarios modelled in 2013, referred to as the Upper Bound and Lower Bound, were plotted near the diffuser, in the main basin and at the outlet of Snap Lake.

The whole-lake average concentrations of TDS and those parameters above AEMP benchmarks were plotted from 2004 to 2013, and were compared to the whole-lake averages predicted in the 2013 modelling update. Temporal trends in whole-lake average predictions were not provided in the EAR; therefore, they were not included in the whole-lake average temporal plots. To provide context for any observed differences between actual and predicted TDS lake concentrations, the actual and predicted TDS loadings from the treated effluent discharge were also reviewed.

Seasonal Kendall Test

For many water quality parameters (e.g., TDS and major ions), temporal trends could be visually identified so rigorous statistical testing was not required to identify differences from the normal range. Where confirmation was required, the Seasonal Kendall Test was used to confirm trends.

The Seasonal Kendall Test was used to remove seasonal cycles and test for the presence of an upward trend, downward trend, or two-sided trend in the data. The test for an upward trend was selected when an increasing trend was visible in the plotted dataset. The test for a downward trend was selected when a decreasing trend was visible in the plotted dataset. The test for a two-sided trend was selected when neither an increasing nor a decreasing trend was visible in the plotted dataset. Statistical significance is obtained from a standard normal distribution for datasets larger than 10. The test generates a z-score (SD) and a *P*-value at a 95% confidence interval. Either the z-score or the *P*-value can be used to evaluate the significance of the trend. SYSTAT 13.1.00.5 was used to complete the statistical analyses in 2013 (SYSTAT 2009). The same stations selected to represent the different lake areas in the visual review for temporal trends were used in the Seasonal Kendall Test: diffuser area (SNAP13 and SNP 02-20e), main basin (SNAP05, SNAP09, and SNAP08), and northwest arm (SNAP02 and SNAP02A).

In 2013, the Seasonal Kendall test was used to test for trends in six parameters (i.e., TP, field pH, manganese, antimony, aluminum, and zinc) for which visual inspection of trends were inconclusive and additional analysis was required to confirm or reject the existence of a trend (Section 3.4.4). The results of the trend analysis for TP was used to support the observation that TP is not increasing in Snap Lake, based on a visual review of temporal plots. Field pH and manganese were tested for trends because increasing trends in the temporal plots were not clearly visible. Total antimony, aluminum, and zinc were above predictions, but temporal trends were not identified through conductivity correlations or visual review of temporal plots; therefore, the Seasonal Kendall test was used to confirm that these three metals were not increasing in Snap Lake.

Dissolved Oxygen

Vertical profiles of DO were also plotted over time to determine whether DO concentrations are decreasing over time at any given depth or within a lake area and, if so, whether the decreases are consistent with EAR predictions.

3.2.2.5 Key Question 4: Are Spatial and Seasonal Patterns in Water Quality in Snap Lake and Downstream Waterbodies Consistent with Predictions Presented in the EAR and Subsequent Modelling Predictions?

Spatial Patterns

Field measurements of conductivity from Snap Lake were used to map the spatial patterns of the treated effluent plume in Snap Lake. Vertical profiles were used to investigate the portion of water column influenced by treated effluent. Additionally, a series of figures showing the plume at snapshots through time were prepared to show both horizontal and vertical spatial patterns of water quality within Snap Lake. For these figures, conductivity between sampling stations was estimated using a discretized thin plate spline interpolation technique (Topo to raster tool in ArcGIS 10.1 using the Spatial Analyst extension), which estimates conductivity values between sampling stations. Conductivity values measured at 12 sampling stations in Snap Lake were used to approximate a smooth thin plate spline surface. The resulting conductivity surfaces minimize surface curvature yielding a smooth surface that passes exactly through the sampling station points (Wahba 1990; Hutchinson 2000; ESRI 2012). The maps presenting near-surface, mid-depth, and near-bottom conductivity values were based on the single field conductivity measured at those depths at each station for May 2013. One map showing mid-depth conductivity was prepared for September 2013 because a vertical conductivity gradient was absent at most stations in Snap Lake in this month.

Based on the QA/QC review of the 2013 field profile data, field conductivity profiles at the diffuser stations in February and field pH profiles measured at AEMP stations between May 7 and May 10, 2013 were invalidated (Appendix 3A, Section 3A.1.2.1). In February, laboratory conductivity values, from samples collected at the bottom, mid-depth, and surface of the lake, were used to show three points within the vertical profiles at the diffuser stations. The pH profile data collected in March 2013 were used, in place of the invalid May pH values, to represent the pH profiles later in the 2013 ice-covered season.

Quality issues found in the February and May 2013 field profiles are discussed in further detail in Appendix 3A.

Seasonal Patterns

Seasonal patterns in key parameters within each of the major parameter groups were identified through plots of average concentrations in different areas of Snap Lake and in the reference lakes. Data from each area in Snap Lake (i.e., diffuser, main basin, and northwest arm), Northeast Lake, and Lake 13 were separated by season (i.e., open-water and ice-covered). Results from the ice-covered season included data collected between November 2012 and May 2013, and open-water results included data collected between July and October 2013.

Downstream Extent of Treated Effluent

The Downstream Lakes Special Study was conducted in three lakes immediately downstream of Snap Lake to delineate the spatial extent of the treated effluent plume and assess current conditions.

A summary of Downstream Lakes Special Study is provided as part of the response to Key Question 4 (i.e., assessing spatial patterns downstream of Snap Lake). Details of the Downstream Lakes Special Study are provided in Section 11.3.

Water quality data from the farthest downstream AEMP station, KING01, were reviewed to identify potential changes in water quality 25 km downstream of Snap Lake. Temporal patterns in TDS and conductivity were reviewed at KING01 to identify trends in TDS. A Seasonal Kendall test for temporal trends was completed to statistically test for trends in TDS at KING01. The annual water quality results at KING01 were compared to AEMP benchmarks and baseline data from KING01 to evaluate the relevance of any increasing trends in TDS at this location.

3.2.2.6 Key Question 5: Is there Evidence of Acidification Effects from the Mine on Nearby Waterbodies?

Water quality data for the three inland lakes (i.e., at stations IL3, IL4, and IL5) were evaluated by comparing mean alkalinity concentrations to a scale presented by Saffran and Trew (1996, Table 3-6), while concentrations of sulphate, nitrate, laboratory and field pH, alkalinity, and base cations were examined for trends which might be indicative of acidification as a result of Mine emissions.

Table 3-6 Acid Sensitivity Scale for Lakes Based on Alkalinity Range

Acid Sensitivity	Alkalinity	
	(mg/L as $CaCO_3$)	(µeq/L)
High	0 to 10	0 to 200
Moderate	>10 to 20	>200 to 400
Low	>20 to 40	>400 to 800
Least	>40	>800

Note: Acid sensitivity scale from Saffran and Trew (1996).
mg/L= milligram per litre; µeq/L = microequivalent per litre; $CaCO_3$ = calcium carbonate; > = greater than.

Water quality data from Streams S1 and S27 were reviewed to identify any changes in stream water quality related to mining activities, including potential acidification effects, and to document loadings to Snap Lake from this source.

3.2.2.7 Key Question 6: Is Water from Snap Lake Safe to Drink?

Parameter concentrations in Snap Lake were predicted to remain below drinking WQGs. Water quality data collected from various locations in Snap Lake, as part of the AEMP, were compared against Canadian drinking WQGs (Health Canada 2012). Canadian drinking WQGs that are health-based are reported as maximum acceptable concentrations (MAC).

Water quality guidelines related to the physical characteristics of the water (i.e., taste, odour, colour) are referred to as aesthetic objectives (Health Canada 2012). Aesthetic objectives (e.g., TDS, iron) were considered in the assessment, as these can influence a user's perception of water drinkability. However, these objectives are not an indication of adverse effects to human health. Although coliform levels were compared to the MAC, the requirement to disinfect all drinking water was considered when reviewing bacteriological results in Snap Lake. Therefore, the 2013 water quality data were compared to both the relevant MAC, including those related to coliforms, and aesthetic objectives.

To provide the required information to assess whether a water quality Action Level was triggered for drinking water protection (Section 3.2.2.8), concentrations at all Snap Lake AEMP stations were compared to 75% of the non-bacteriological drinking WQGs (Health Canada 2012), and 75% of all wildlife health WQGs were based on livestock watering guidelines (CCME 1999).

Water quality data collected at the raw water intake station (SNP 02-15) were also compared to drinking WQGs. The water quality at the raw water intake, along with the location and frequency of values above drinking water guidelines in Snap Lake, were used to assess whether water quality in Snap Lake was safe to drink.

3.2.2.8 Action Level Assessment

The 2013 AEMP Design Plan (De Beers 2014 includes a Response Framework that provides a systematic approach to responding to the AEMP monitoring results such that the potential for significant adverse effects is identified and any necessary mitigation actions are undertaken. Action Levels were defined for key values or Assessment Endpoints in Snap Lake. For water quality, Action Levels are related to protecting the following values: ecological function of Snap Lake maintained; and, water in Snap Lake is safe to drink.

The water quality Action Levels related to protecting the ecological function of Snap Lake were separated into two potential effects from the Mine identified in the EAR: toxicological impairment, and nutrient enrichment. The water quality Action Levels related to protecting the ecological function of Snap Lake, both from a toxicological impairment and nutrient enrichment perspective, and maintaining safe drinking water in Snap Lake are summarized in Table 3.7. Responses to the key questions identified in Table 3-5 were compiled to assess whether any action levels have been triggered. Additional details regarding the assessment of water quality action levels are provided in the 2013 Design Plan (De Beers 2014a).

Table 3-7 Summary of Water Quality Action Levels

Value		Key Information Used in Assessing Action Levels	Negligible	Low Action Level
Ecological Function Maintained	Toxicological Impairment	Differences between Snap Lake and reference lakes or normal range; AEMP Benchmarks	Concentration not exceeding AEMP Benchmarks where they exist, or if exceeding, not due to Mine (KQ2) **AND** Within normal range lake-wide (KQ3)	Concentration greater than normal and reference range lake-wide supported by a temporal trend (KQ3) **AND** Exceeding 75% of AEMP Benchmark[a] at the edge of the mixing zone (i.e., at diffuser stations) (KQ2)
		Toxicity results near edge of mixing zone	No persistent sublethal toxic effects to test organisms in mixing zone samples (KQ2)	Persistent sublethal toxic effects to test organisms in mixing zone samples (KQ2) **OR** Sublethal toxic effects for Fish Early Life Stage test in mixing zone samples (KQ2)
	Nutrient Enrichment	Differences between Snap Lake and reference lakes or normal range; AEMP Benchmarks and site-specific benchmarks	Consistent with EAR prediction (KQ3) **AND** If AEMP Benchmark exists, below the benchmark (KQ2)	Exceeding EAR Predictions supported by temporal trend (KQ3) **AND** Exceeding >75% AEMP Benchmark, if it exists (KQ2)
Water must be drinkable		Drinking Water and Wildlife Health Guidelines	Drinking water parameters <75% Health Canada human health and aesthetic drinking WQG (KQ6) **AND** Microcystin-LR <75% of Health Canada human health drinking WQG (KQ6) **AND** Drinking water parameters <75% CCME wildlife health WQG (KQ6)	Drinking water parameters at any location are above 75% of Health Canada human health or aesthetic drinking WQG (KQ6) **OR** Microcystin-LR at any location is above 75% of Health Canada human health drinking WQG (KQ6) **OR** Drinking water parameters at any location are above 75% of CCME wildlife health WQG (KQ6)

a) Benchmarks currently used in the AEMP to which substance concentrations are compared (i.e., EAR benchmarks and CCME guidelines).

AEMP = Aquatic Environmental Monitoring Program; KQ = key question; < = less than; > = greater than; % = percent; CCME = Canadian Council of Ministers of the Environment; WQG = water quality guideline; LR = lysine-arginine.

3.3 Quality Assurance and Quality Control

3.3.1 Overview of Procedures

Quality assurance and quality control (QA/QC) procedures govern all aspects of the AEMP (i.e., field methods, laboratory analysis, data management and analysis, and reporting). Field QA/QC procedures pertain to the maintenance and operation of equipment and instrumentation, sampling methods, sample handling, and shipping. Laboratory QA/QC procedures incorporate protocols developed by analytical laboratories. Office QA/QC procedures involve validation of field measurements and analytical results provided by analytical laboratories. Details of QA/QC procedures specific to the AEMP are provided in the De Beers QA/QC Plan (De Beers 2008b) and in the QA/QC Procedures and Results for the Water Quality Program (Appendix 3A). The results of the 2013 QA/QC program are summarized in this section.

3.3.2 Summary of Results

3.3.2.1 Qualified Data

Data were qualified if holding times were exceeded or parameter concentrations in the field, trip, or equipment blanks were similar to those measured in the lake. In 2013, approximately 25% of the laboratory data were qualified, consistent with previous years. Qualified data were flagged with these abbreviations:

- WH: warning, holding time was exceeded and may have an effect on results;
- NP: lake patterns using this result should be reviewed because parameter concentrations in either the equipment, field, or travel blanks were above detection levels and at or near lake concentrations, and occurred at a moderate to high frequency; and,
- QP: lake patterns using this result should be reviewed because parameter concentrations in the equipment, field, or travel blanks were above the detection limit, at or near lake concentrations, and occurred at a low to moderate frequency.

Data with WH, NP, and QP qualifiers made up 11%, 10%, and 3% of the dataset, respectively. All data with those qualifiers were used in the water quality assessment in this AEMP. However, the qualifiers were considered further when data showed a potential pattern or were above an AEMP benchmark, EAR prediction, or drinking water guideline.

The percentage of ALS samples that exceeded warning holding times ranged from 1% for F2 (>C_{10} –C_{16}) to 100% for laboratory pH. nitrite, nitrate, nitrite plus nitrate, and ortho-phosphate often exceeded warning holding times (99%).

Thirteen parameters were qualified with “NP” and three parameters were qualified with “QP”. The number of flagged parameters in blank samples was lower in 2013 compared to 2012. The parameters flagged with “NP” were turbidity, TOC, total and dissolved aluminum, antimony, boron, copper, zinc, and dissolved manganese. The parameters qualified with the “QP” flag were reactive silica, total lead, and total nickel.

Additional evaluation of potential contamination of total and dissolved antimony in the blank samples confirmed that antimony contamination likely occurred in the 2013 AEMP lake samples (i.e., lake concentrations of antimony were likely over-estimated) (Appendix 3A). Therefore, total and dissolved antimony results should be reviewed with this consideration in mind. For all other flagged parameters, results indicate that contamination was either isolated to the blanks or at concentrations low enough relative to Snap Lake data to allow for adequate interpretation of the data in terms of guideline comparisons and spatial and temporal patterns.

The details of QA/QC methods and results for the 2013 AEMP water quality program are presented in Appendix 3A.

3.3.2.2 Invalidated Data

Field Data

Approximately 3% of the field data were invalidated because:

- The measurement probes were assumed to be near the sediment boundary, or submerged in sediment as denoted by substantial changes in DO, associated with notable changes in pH, temperature, and/or conductivity values.
- Dissolved oxygen as percent saturation (% sat) was inconsistent with the measured DO value.
- Field conductivity measurements were inconsistent with spatial patterns typically observed in Snap Lake, and were different from laboratory conductivity results from the same stations.
- Calibration failure was noted in the calibration log for field measured pH.

Laboratory Data

In 2013, less than 1% of the laboratory data were invalidated. Data that were invalidated were flagged with an 'X' in the De Beers Environmental Database and were not used in the analyses.

Specific anomalous data points were also removed, based on the criteria outlined in Appendix 3A, Section 3A.1.2.3. In 2013, the case-by-case values that were invalidated were:

- ions: fluoride collected at the diffuser station SNP 02-20f on May 7, 2013; and,
- total metals:
  - lead from SNAP23 collected on May 9, 2013;
  - zinc from SNAP09 collected on Feb 9, 2013; and,
  - zinc from SNAP02A collected on September 5, 2013.

These invalidated data were removed because they were considered unusually low or high results. In each case, the concentrations were more than ten times lower or higher than the average concentrations measured in Snap Lake during the 2013 sampling period. Additional detail is provided in Appendix 3A.

In 2013, none of the results were invalidated based on holding time exceedances because all samples met the holding time frames that would likely affect the result in 2013 (i.e., ten days after the holding time had expired); however, most results that exceeded holding times were given the WH qualifier as outlined in Section 3.3.2.1.

Data were also invalidated due to occurrences of dissolved metal concentrations higher than the total metals concentration in the same sample. Less than 1% of dissolved metals results were invalidated because the relative percent difference (RPD) was higher than 30% between corresponding dissolved and the total metals concentrations.

Overall, the number of parameters that failed to comply with QC criteria was low compared to the total number of parameters analyzed. Therefore, the quality of water quality data collected during the 2013 AEMP was considered acceptable and adequate to address the objectives of the monitoring program.

3.4 Results

3.4.1 Summary of Snap Lake Water Quality

This section provides a high-level summary of water quality in Snap Lake in 2013. Specific key questions are answered in Sections 3.4.2 to 3.4.7 and provide detailed rationale for the patterns and trends outlined below.

Snap Lake is relatively shallow, with a mean depth of approximately 5 m, and is well-mixed during open-water conditions, with the exceptions of one deeper diffuser station (SNP 02-20e) and a deeper area in the northwest arm (greater than 20 m deep), where thermoclines have been observed. During ice-covered conditions, limited mixing occurs in Snap Lake. Snap Lake is clear, as indicated by a Secchi depth greater than 6 m, and is acidic to alkaline in terms of pH.

Concentrations of DO during ice-covered conditions in 2013 in Snap Lake tended to be near saturation at the surface, immediately under the ice, and decreased with depth. This pattern was also observed in Snap Lake under baseline conditions (De Beers 2002), and is consistent with observations in Northeast Lake (Section 3.4.5). Since treated effluent discharge to Snap Lake began, the expected decline in DO concentrations during ice-covered conditions in deep waters in the main basin of Snap Lake has not occurred. Additionally, minimum DO concentrations have been higher than measured during the baseline period.

In 2013, alkalinity in Snap Lake ranged from 10 to 34 mg/L as calcium carbonate ($CaCO_3$), which indicates a high to low sensitivity to acidification (Saffran and Trew 1996). However, due to increasing alkalinity and pH in Snap Lake since discharge of treated effluent began, the lake is becoming less sensitive to acidification. Increasing lake alkalinity concentrations, which are consistent with the elevated alkalinity in the treated effluent relative to baseline, lower the potential for acidification by increasing the buffering capacity of the lake. Both field and laboratory pH values are increasing in Snap Lake.

Under baseline conditions in Snap Lake, the dominant ions were calcium and bicarbonate. Since discharge to Snap Lake began, the relative proportion of the bicarbonate anion has decreased, while the relative proportion of the chloride anion has increased. The major ionic composition in Snap Lake is shifting to closely reflect the ionic composition of the treated effluent (e.g., calcium and chloride), which is expected because the treated effluent discharge is the major source of major ions to Snap Lake.

Total hardness has increased in Snap Lake since discharge of treated effluent began, thereby lowering the potential for metals toxicity. Total hardness concentrations increased from 37 mg/L in 2004 to 164 mg/L in 2013 at the outlet of Snap Lake (SNAP08). Increases in total hardness of lake water reduce the potential toxicity of some metals (Chapman 2008).

Baseline TP concentrations in Snap Lake indicated low to moderate productivity, or an oligotrophic to lower mesotrophic status (De Beers 2002). Increasing concentrations of nutrients are expected in areas influenced by the treated effluent discharge, because treated effluent contains elevated concentrations of nitrogen and phosphorus. Overall, there have been no clear temporal trends in TP concentrations since 2004, but nitrate and ammonia concentrations have increased in Snap Lake. As expected, higher concentrations of nitrate and ammonia were observed at stations closest to the diffuser.

Phosphorus was determined to be the limiting nutrient in Snap Lake (De Beers 2002) because the nitrogen-to-phosphorus ratio for waters in both the main basin of Snap Lake and the northwest arm was greater than 23 to 1. A nitrogen-to-phosphorus ratio of 23 to 1 is the lower boundary of a *P*-limited system (Wetzel 2001). Given the measured annual increases of ammonia and nitrogen concentrations in the main basin of Snap Lake relative to phosphorus, phosphorus continues to be the limiting nutrient.

Water quality parameters in Snap Lake were below AEMP benchmarks and Water Licence limits with the exception of chloride, fluoride, and nitrate. Concentrations of chloride, fluoride, and nitrate in Snap Lake were low during baseline conditions, but were predicted to increase as a result of treated effluent discharge (De Beers 2002). Concentrations of these parameters remained below predicted concentrations and recommended SSWQOs for Snap Lake.

Nine metals, which are present in elevated concentrations in the treated effluent discharge, are increasing in Snap Lake: barium, strontium, boron, lithium, manganese, nickel, rubidium, uranium, and molybdenum. Concentrations of these metals were lower in the northwest arm of Snap Lake than in other areas of the lake. This pattern was expected since the northwest arm is isolated from the discharge area compared to other areas of Snap Lake.

3.4.2 Key Question 1: Are Concentrations or Loads of Key Water Quality Parameters in Discharges to Snap Lake Consistent with EAR Predictions and Below Water Licence Limits?

3.4.2.1 Inputs to Snap Lake

In 2013, inputs to Snap Lake from Mine-related activities included treated effluent discharged through one or two diffusers and uncontrolled runoff. The term “treated effluent” refers to combined treated water from the WTP and TWTP. The diffusers do not influence total loadings to Snap Lake or lake-wide changes in water quality. Each diffuser maximizes initial mixing of the treated effluent discharged to Snap Lake and can reduce TDS concentrations and concentrations of other constituents near the diffuser. The term “uncontrolled runoff” refers to water that collects in bogs and catchments that may enter Snap Lake. These runoff areas are monitored as part of the SNP. Since treated effluent is the major contributor to water quality in Snap Lake, the quality and quantity of treated effluent are discussed in Section 3.4.2.2. Runoff volumes from all the surface runoff locations were small compared to the volume of Snap Lake; therefore, changes in water quality in Snap Lake are expected to be localized, temporary, and negligible relative to changes resulting from the treated effluent plume. Uncontrolled runoff is discussed in the 2013 ARD and Geochemistry Report, located in Appendix A of the Water Licence Annual Report submitted to MVLWB in accordance with the Water Licence (De Beers 2014c).

3.4.2.2 Treated Effluent

Discharge of treated effluent to Snap Lake from Mine dewatering activities began on June 22, 2004 using a temporary diffuser. Key modifications to discharge location and/or treated effluent composition are outlined for each subsequent monitoring year below:

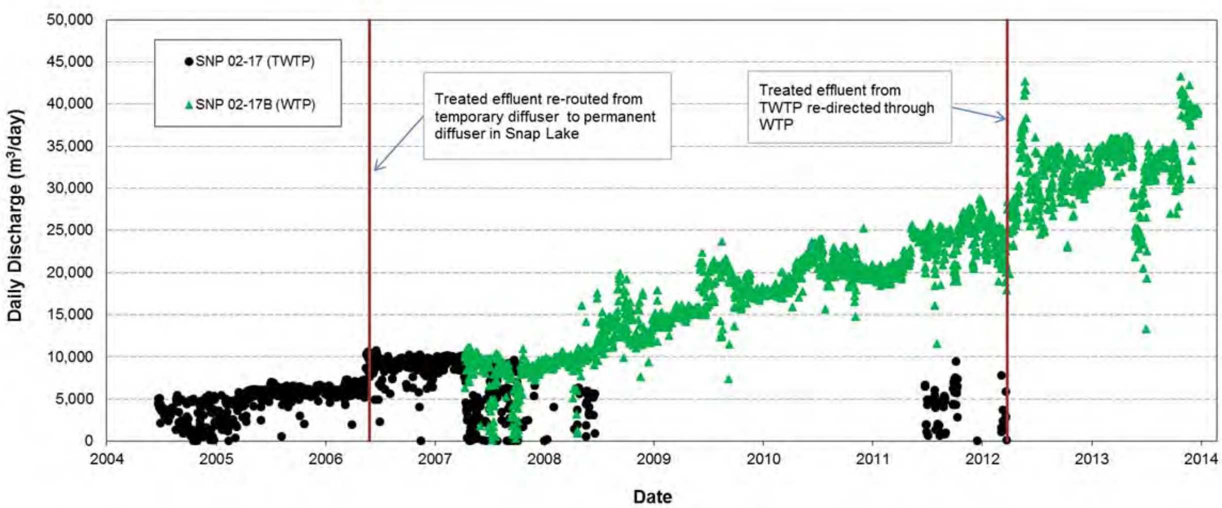
- May 29, 2006, the treated effluent was re-routed from the temporary diffuser to the permanent diffuser.
- In 2007, most of the treated effluent was discharged through the permanent diffuser.
- In 2008, most of the treated effluent was routed through the WTP, with smaller volumes routed through a TWTP. All of the 2008 treated effluent was directed through the permanent diffuser, with the exception of a small volume of treated domestic waste water (approximately 0.1%), which was released to the wetlands near the northwest arm.
- In 2009 and 2010, the WTP treated all treated effluent. All domestic waste water was treated and routed through the WTP, with the exception of a one-day discharge from the domestic waste water treatment plant to the wetlands near the northwest arm in 2009, which was the last time domestic waste water was released to the wetlands..
- In 2011, most of the treated effluent was routed through the WTP, with smaller volumes routed through a TWTP periodically between June and the end of October. All of the 2011 treated effluent was directed through the permanent diffuser.

- The permanent diffuser was replaced with a new diffuser in September 2011, herein referred to as the 2011 permanent diffuser.
- In 2012, most of the treated effluent was routed through the WTP, with smaller volumes routed through the TWTP until March.
- In March 2012, the treated effluent from the TWTP was redirected through the WTP, after the junction of the water management line from the TWTP was relocated upstream of the WTP. Combined treated effluent was discharged from the WTP to Snap Lake through the 2011 permanent diffuser.
- During the spring freshet of 2012, a temporary floating diffuser was installed on the ice near the 2011 permanent diffuser, in accordance with the approved Freshet Water Management Plan (Golder 2012a). Combined treated effluent was discharged from the WTP through both the temporary floating diffuser and the 2011 permanent diffuser between May 20, 2012 and June 5, 2012.
- In 2013, combined treated effluent from the WTP was discharged through the 2011 permanent diffuser until May 17, 2013, and then from May 18, 2013 to October 5, 2013, through both the temporary floating diffuser and the 2011 permanent diffuser.
- In August 2013, the modification notice regarding the installation of a second permanent diffuser was approved (MVLWB 2013d). The temporary floating diffuser was decommissioned when the second permanent diffuser, herein referred to as the 2013 permanent diffuser, started operating on October 6, 2013. As of October 6, 2013, treated effluent was discharged to Snap Lake through both the 2011 and 2013 permanent diffusers.

Quantity

Approximately 14 million cubic metres (Mm^3) of combined treated effluent was discharged from the WTP into Snap Lake through one or two diffusers between January 2013 and December 2013 (Section 2, Table 2-6). The total discharge volume was approximately 31% higher than in the 2012 AEMP reporting year. Discharge flows have increased over time since the start of the Mine in 2004 (Figure 3-4).

Figure 3-4 Treated Effluent Discharge Rate to Snap Lake, 2004 to December 2013



Note: The daily discharge rate in 2013 shows total discharge from the permanent water treatment plant (i.e., measured at SNP 02-17B), which included discharges, at various times, from the 2011 permanent diffuser, the temporary floating diffuser, and the 2013 permanent diffuser.

TWTP = temporary water treatment plant; WTP = permanent water treatment plant; m^3/day = cubic metres per day.

Quality

Comparisons to Water Licence Limits

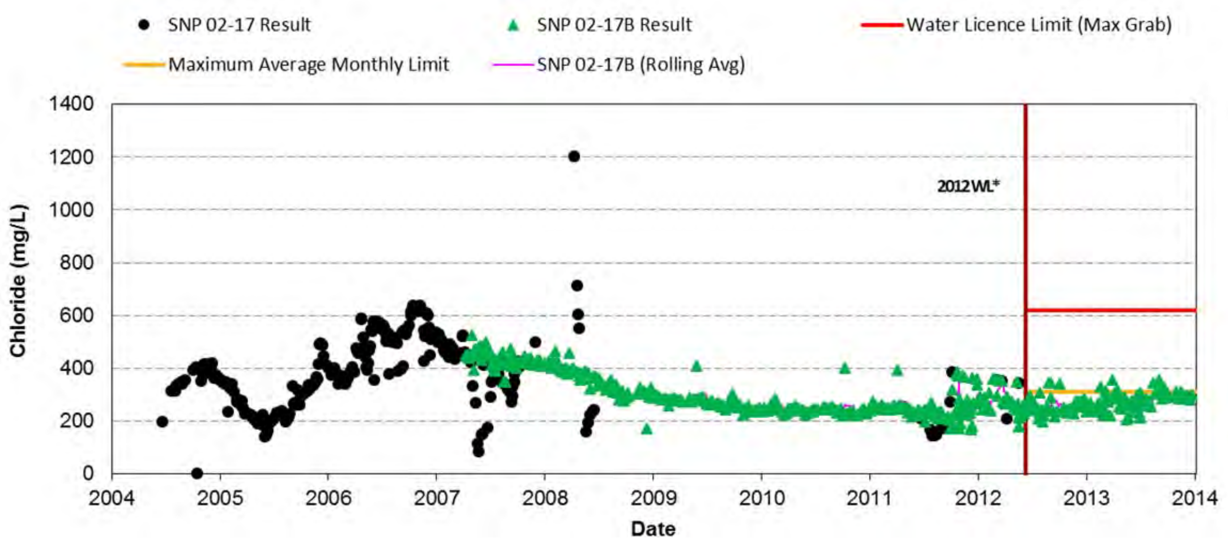
Parameter concentrations in the treated effluent were below both the maximum concentration in any grab sample and the average monthly limit (AML) between January 2013 and December 2013, with the exception of chloride concentrations, which were above the AML of 310 mg/L in September 2013 (Figure 3-1). As documented in the SNP reports that discussed the chloride exceedances (De Beers 2013d,e,f,g), additional treated effluent samples were collected from October 2013 to December 2013 to confirm chloride concentrations decreased below the AML.

Chloride is a major component of TDS, which is also increasing in Snap Lake due to treated minewater discharges. In response to increasing TDS concentrations in Snap Lake, De Beers prepared the TDS Response Plan, which includes management options for TDS and, by extension, chloride and other major ions. The TDS Response Plan evaluates existing and future management practices to maintain TDS and chloride concentrations at acceptable concentrations in Snap Lake (De Beers 2013h). Site-specific water quality objectives for TDS and chloride were also developed in the TDS Response Plan to better define acceptable concentrations for TDS and chloride in Snap Lake.

Plots for all other parameters analyzed in the discharge are provided in Appendix 3E.

Figure 3-5 Chloride Concentrations in Treated Effluent, 2004 to December 2013

a. Concentration and Rolling Average

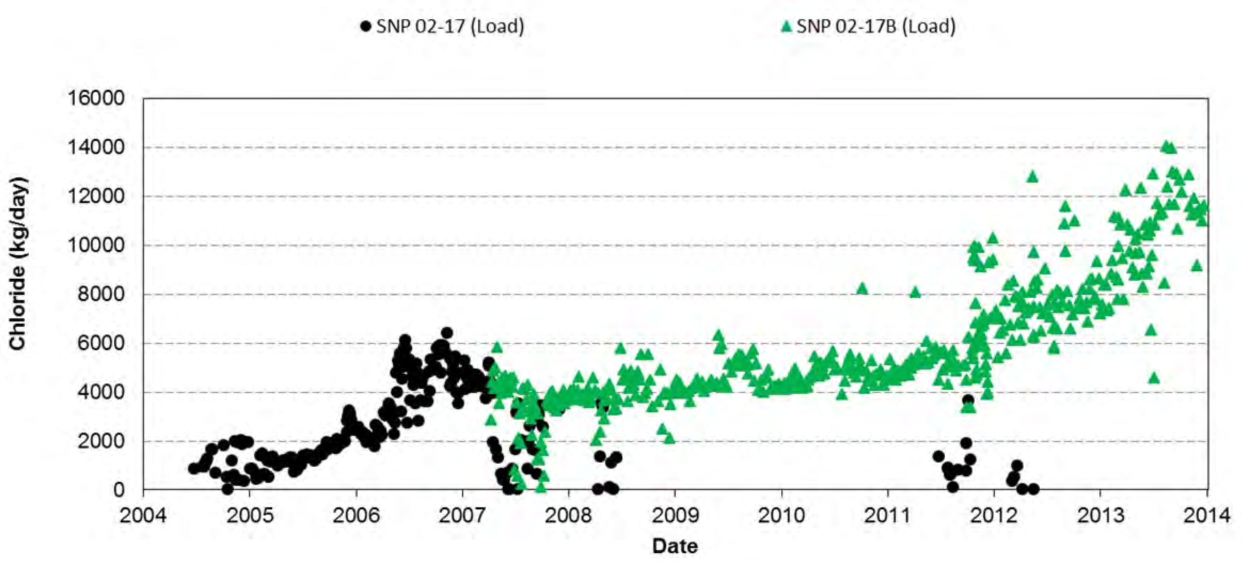


* The Water Licence limits (maximum concentration of any grab sample and maximum average monthly limit) were effective on June 14, 2012: MV2011L2-0004 (MVLWB 2012).

Rolling Avg = a rolling average calculated based on the letter from MVLWB (2013c); SNP 02-17 = treated effluent from the temporary water treatment plant; Max Grab = maximum allowable concentration in any grab sample; SNP 02-17B = treated effluent from the permanent water treatment plant;

mg/L = milligrams per litre.

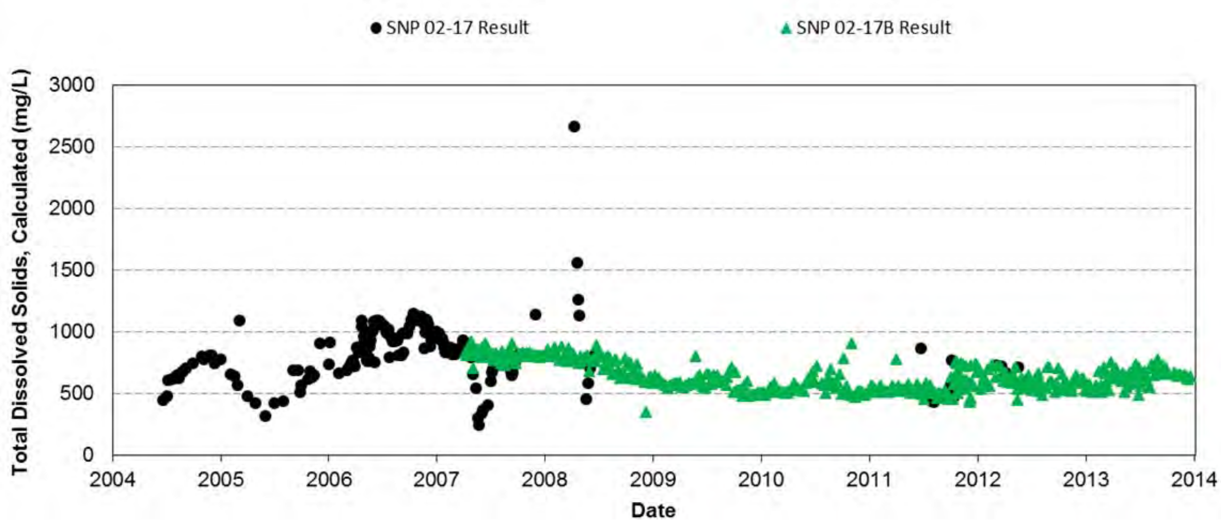
b. Loading



SNP 02-17 = treated effluent from the temporary water treatment plant; SNP 02-17B = treated effluent from the permanent water treatment plant; kg/day = kilograms per day.

Figure 3-6 Total Dissolved Solids Concentrations in Treated Effluent, 2004 to December 2013

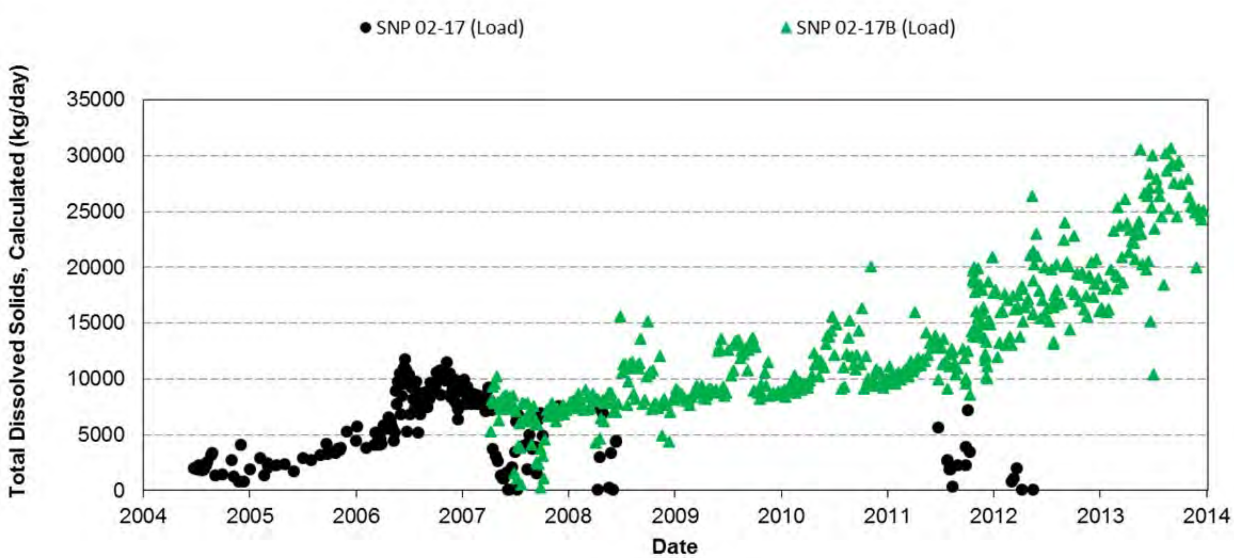
a. Concentration



Note: TDS concentrations were calculated based on the formula described in Part 1030 E in the Standard Methods for the examination of water and wastewater (APHA 2005).

SNP 02-17 = treated effluent from the temporary water treatment plant; SNP 02-17B = treated effluent from the permanent water treatment plant; mg/L = milligrams per litre.

b. Loading



SNP 02-17 = treated effluent from the temporary water treatment plant; SNP 02-17B = treated effluent from the permanent water treatment plant; kg/day = kilograms per day.

Total Phosphorus, Ammonia, and Nitrate Loadings to Snap Lake

Based on the average flow-weighted concentrations of TP, the total loading of TP to Snap Lake was approximately 62 kilograms (kg) between January 1, 2013 and December 31, 2013 (De Beers 2014c). The Water Licence specifies an annual TP load of less than 256 kg/y to Snap Lake (MVLWB 2013a); thus, loadings were well below the Water Licence limit. Similarly, annual loadings of ammonia (16,797 kg/y) and nitrate (86,765 kg/y) were also well below the annual limits of 187,000 kg/y and 219,000 kg/y, respectively.

Comparisons to Environmental Assessment Report Predictions and 2013 Modelling Predictions

The flow-weighted average concentrations of most parameters were below maximum annual average concentrations predicted for treated effluent in the EAR, with the exception of sulphate (Table 3-8). The flow-weighted average concentration for sulphate has consistently been above the EAR prediction since 2004 (De Beers 2011, 2012b, 2013a). However, the sulphate flow-weighted average concentration was within the range of maximum average annual sulphate concentration predicted in the 2013 modelling update (i.e., 49 mg/L to 69 mg/L; De Beers 2013b). Sulphate is a component of TDS (i.e., approximately 9%), so it was implicitly considered as part of the aquatic toxicity testing conducted to develop an appropriate SSWQO for TDS (De Beers 2013h).

The flow-weighted average concentrations were below or similar to (i.e., within 10% of) the upper end of the range in predicted average concentrations from the 2013 modelling update, with the exception of aluminum, chromium, iron, and lead. Differences between the 2013 flow-weighted average concentrations and the 2013 model predictions could be related to uncertainties in modelled groundwater inflows; the heterogeneity in underground fractures results in uncertainties in predictions of groundwater inflows and concentrations (De Beers 2013b).

Table 3-8 Summary of Comparisons between Observed and Predicted Treated Effluent Discharge Concentrations in 2013

Parameters	Units	SNP 02-17B				EAR Predictions[b]			Range in 2013 Average Predictions[c]
		Min	Weighted Average[a]	Max	Count	Average	Maximum Average Annual	Maximum Weekly	
Field Measurements									
Total Dissolved Solids, calculated (Lab)[d]	mg/L	484	*632*	778	69	592	929	1,332	674 - 971
Total Suspended Solids	mg/L	<3	2	6	70	5	5	5	-
Major Ions									
Calcium	mg/L	97	128	176	70	153	235	558	139 - 203
Chloride	mg/L	204	*278*	354	70	237	374	425	297 - 437
Fluoride	mg/L	0.25	0.33	0.45	70	-	-	-	0.39 - 0.40
Magnesium	mg/L	11	<u>16</u>	22	70	16	21	25	15
Potassium	mg/L	4	<u>5</u>	8	70	12	16	17	5
Reactive Silica, as SiO_2	mg/L	7	9	11	70	-	-	-	9 - 10
Sodium	mg/L	46	*64*	80	70	38	69	78	77 - 111
Sulphate	mg/L	42	***55***	67	70	17	40	46	49 - 69
Nutrients									
Nitrate, as N, calculated	mg-N/L	2.0	*6.7*	16.3	70	5.8	13.3	15.8	9.4 - 9.6
Ortho-phosphate	mg-P/L	<0.001	0.001	0.0023	70	0.008	0.011	0.023	0.0021
Total Kjeldahl Nitrogen	mg-N/L	0.5	1.3	3.1	70	6.8	8.6	9.3	1.5
Total Metals									
Aluminum	µg/L	10	<u>28</u>	60	72	<100	<100	<100	16
Barium	µg/L	29.7	<u>38</u>	47	72	337	416	437	35 - 36
Chromium	µg/L	0.15	<u>0.44</u>	1.06	72	7.46	7.49	7.51	0.26
Cobalt	µg/L	0.14	<u>0.32</u>	0.63	72	0.60	3.15	3.40	0.3 - 0.31
Iron	µg/L	36	<u>88</u>	272	72	<300	<300	<300	31
Lead	µg/L	0.05	<u>0.14</u>	0.25	72	0.73	0.93	9.20	0.04
Manganese	µg/L	44	*62*	90	72	30	146	156	63 - 64
Molybdenum	µg/L	3.1	5.4	9.7	72	8.4	10.0	79.9	5.7
Nickel	µg/L	10	<u>13</u>	17	72	14	15	61	13

Table 3-8 Summary of Comparisons between Observed and Predicted Treated Effluent Discharge Concentrations in 2013

Parameters	Units	SNP 02-17B				EAR Predictions[b]			Range in 2013 Average Predictions[c]
		Min	Weighted Average[a]	Max	Count	Average	Maximum Average Annual	Maximum Weekly	
Total Metals (Continued)									
Strontium	µg/L	1,270	*1,769*	2,350	72	1,501	2,346	2,616	1,560 - 2,259
Thallium	µg/L	<0.01	0.01	0.02	72	0.12	0.13	0.36	0.01
Uranium	µg/L	0.67	*<u>1.03</u>*	1.46	72	0.68	1.17	17.71	0.96 - 0.98
Vanadium	µg/L	<0.05	0.13	0.26	72	2.30	3.12	43.90	0.21
Zinc	µg/L	1.19	3	11	72	14	17	22	3.77 - 3.78

Note: *Italics* indicates a flow-weighted average concentration above the predicted average from the EAR; **bold** indicates a weighted-average concentration above the predicted maximum annual average from the EAR; **bold** and *italics* indicate a weighted-average concentration above the predicted weekly maximum average from the EAR; <u>Underline</u> indicates a weighted-average concentration above the range in 2013 prediction averages.

a) The flow-weighted average was calculated using the daily discharge for each sample; non-detectable results were set to half the detection limit.

b) EAR predictions from De Beers (2002).

c) The range was the annual average concentrations using predicted daily concentrations for treated effluent in 2013 from the four modelled scenarios (i.e., Upper Bound Scenarios A and B and Lower Bound Scenarios A and B (De Beers [2013b]). Values in the table were rounded to the same level of precision as measured values after the flow-weighted average concentration and predictions were compared (Appendix 3E). Single values indicate that upper and lower end of the range were equivalent at the precision of the measured data.

d) "Total dissolved solids, calculated (Lab)" refers to laboratory-calculated total dissolved solids concentrations using a formula adapted from the Method 1030 E in the Standard Methods for the Examination of Water and Wastewater (APHA 2005). Refer to Appendix 3A for further details.

Min = minimum; Avg = average; Max = maximum; EAR = Environmental Assessment Report; SiO_2 = silicate; N = nitrogen; mg/L = milligrams per litre; µg/L = micrograms per litre; <= less than the detection limit; SNP = Surveillance Network Program; N = nitrogen; mg-N/L = milligrams as nitrogen per litre; mg-P/L = milligrams as phosphorus per litre.

Toxicity of Discharge

Acute and chronic toxicity tests were conducted on treated effluent samples on a quarterly basis.

The 2013 treated effluent samples did not show any acute toxicity response for either Rainbow Trout or *Daphnia magna*. The regulatory requirement to demonstrate an absence of acute toxicity to juvenile Rainbow Trout (MVLWB 2013a) was confirmed. Acute toxicity has not occurred in any of the treated effluent samples collected from 2005 to 2013.

Chronic toxicity was predicted to occur in treated effluent in the EAR (De Beers 2002). In 2013, the May and October treated effluent samples from the WTP showed evidence of chronic toxicity in terms of *Ceriodaphnia dubia* survival and reproduction. None of the treated effluent samples showed evidence of chronic toxicity in terms of algal growth inhibition. In contrast, most of the algal tests performed on treated effluent showed growth stimulation.

The temporal evaluation of chronic toxicity from 2005 to 2013 demonstrated that, although toxicity did occur in two chronic tests performed on the treated effluent, it did not show a temporal trend of increasing frequency or severity over time.

Details of the toxicity test methods and of the results for the treated effluent samples are provided in Appendix 3F, including graphical summaries of the chronic toxicity data.

3.4.2.3 Dilution Factors for Discharged Minewater Effluent

The minimum dilution factors at 200 m away from the 2011 permanent diffuser (i.e., the edge of the mixing zone) in February, May, July, August, and September 2013 were 16, 28, 7, 25, and 31, respectively (Table 3-9). These minimum dilution factors are based on the maximum observed TDS concentrations at the diffuser stations, which represent the least amount of dilution that was provided by the operating diffuser(s).

Dilution provided by the 2011 diffuser was anticipated to be similar or more than the dilution predicted in the EAR based on the modelling results from the 2013 plume characterization study (Golder 2013), and the steps De Beers took to minimize air in the discharge in September 2012 (De Beers 2012c), which were expected to increasing mixing (De Beers 2012b). The February 2013 dilution factor (16) was similar to the EAR prediction for ice-covered conditions (12) after the initial seven years of operations (De Beers 2002). The May dilution factor (28) was higher than EAR predictions. The 2013 seasonal patterns observed in the dilution factors were consistent with the 2012 seasonal pattern in dilution factors (De Beers 2013a).

The lowest calculated dilution factors in 2013 occurred during the early open-water season in July. The lower dilution factor observed in July was influenced by the presence of a thermocline at one diffuser station (i.e., SNP 02-20e) for a short time during the early open-water season. The July 2013 dilution factor was as low as the 2012 July dilution factor. This was consistent with the results of the modelling completed for the 2013 Plume Characterization study, which indicated that open-water conditions may be the most limiting time for mixing in that area (Golder 2013). The lower DFs are the result of a larger difference between the maximum concentration at the diffuser stations and the background concentration in Snap Lake in July than at other times in the year. The highest observed dilution factor occurred during the late open-water season in September, when the treated effluent dilution appeared to be most affected by additional mixing due to wind-driven currents.

The dilution factors in July and September represent the effective dilution provided by both the temporary floating and 2011 diffusers. The temporary floating diffuser was operating between May 18 and October 5, 2013 and this may have influenced dilution of the effluent as flows from the floating diffuser represented a smaller portion of the total effluent discharge flows (approximately 6 to 38%). As a result, this would have less influence on the characterization of the effluent plume relative to the 2011 diffuser.

In October 2013, a second permanent diffuser (i.e., the 2013 diffuser) replaced the temporary floating diffuser. The 2013 permanent diffuser, which has the same general configuration as the 2011 permanent diffuser, was installed approximately 25 m away in parallel with the 2011 permanent diffuser. The potential changes in mixing characteristics and resulting dilution factors due to the side-by-side configuration of the diffusers will be assessed in the 2014 Plume Characterization study.

Table 3-9 Dilution Factors of the Effluent Discharge Based on 2013 Total Dissolved Solids Results

Month	Average Discharge Rate (m^3/d)	Discharge Range (m^3/d)		Calculated Total Dissolved Solids Concentrations (mg/L)			Minimum Dilution Factor
				Snap Lake Background (Average of Near-Field Stations) [(a)]	Maximum at any Depth at Diffuser Stations[(b)]	Treated Effluent (Flow-Weighted Average)[(c)]	
February	33,732	Min:	30,806	257	278	593	16
		Max:	35,079				
May	39,201	Min:	33,552	289	300	598	28
		Max:	47,596				
July	38,753	Min:	21,376	219	278	620	7
		Max:	44,399				
August	40,199	Min:	33,546	231	248	649	25
		Max:	42,239				
September	39,559	Min:	28,613	241	256	704	31
		Max:	43,191				

Note: For February and May 2013, the minimum dilution factors were representative of dilution prior to discharge from the temporary floating diffuser. For July to September 2013, the minimum dilution factors were representative of dilution with the operation of the temporary floating diffuser.

a) Average TDS concentrations measured at SNAP03, SNAP05, SNAP06, SNAP12, SNAP26, and SNAP28 in February and at SNAP 03, SNAP 05, and SNAP06 in May 2013.

b) Maximum of TDS concentrations that were measured at any depth of diffuser stations (i.e., SNP 02-20d, SNP 02-20e, SNP 02-20f) in 2013.

c) Flow-weighted average TDS concentration from the permanent (SNP 02-17B) WTP in 2013.

Min = minimum; Max = maximum; m^3/d = cubic metres per day; mg/L = milligrams per litre; WTP = water treatment plant.

3.4.2.4 Summary of Key Question 1

The volume of daily discharge to Snap Lake has increased since 2004. Concentrations in the treated effluent were below the maximum allowable concentration in any grab sample of treated effluent for all parameters in 2013. The monthly rolling averages were below the AML, with the exception of chloride concentrations, which were above the AML of 310 mg/L in September 2013. De Beers discussed the chloride exceedance with the Inspector from Aboriginal Affairs and Northern Development Canada and follow-up is ongoing. De Beers has developed a TDS Response Plan, which outlines a process to address concerns related to elevated chloride concentrations in the effluent (De Beers 2013h). The TDS Response Plan includes identifying Mine operations practices to reduce TDS and chloride in the effluent, reviewing effluent treatment options, and using a SSWQO to develop more applicable discharge limits. The 2013 TP loading to Snap Lake from the WTP was 62 kg, which was well below the Water Licence limit of 256 kg.

The 2013 annual flow-weighted average concentrations of sulphate were above the EAR prediction, consistent with previous years. The sulphate flow-weighted average concentration was within the range of maximum average annual sulphate concentration predicted in the 2013 modelling update (De Beers 2013b). The flow-weighted average concentrations in 2013 were below or similar to the upper bound in the 2013 model predictions, with the exception of aluminum, chromium, iron, and lead. The results are likely associated with uncertainties in the model predictions.

The 2013 treated effluent samples did not show any acute toxicity response for either Rainbow Trout or *Daphnia magna*. The regulatory requirement to demonstrate an absence of acute toxicity to juvenile Rainbow Trout (MVLWB 2004, 2012) was confirmed. In 2013, two treated effluent samples from the permanent WTP showed evidence of chronic toxicity in terms of *Ceriodaphnia dubia* survival and reproduction. None of the treated effluent samples showed evidence of chronic toxicity in terms of algal growth inhibition. However, most of the algal tests performed on treated effluent showed growth stimulation.

3.4.3 Key Question 2: Are Concentrations of Key Water Quality Parameters in Snap Lake below AEMP Benchmarks and Water Licence Limits?

3.4.3.1 AEMP Benchmarks

Water quality parameters in Snap Lake were below AEMP benchmarks with the exception of chloride, fluoride, and nitrate (Table 3-10). Dissolved oxygen concentrations and field pH values were, on occasion, below the minimum CCME WQG. For these five parameters, the relevance of these results and the potential risks to aquatic biota are discussed in more detail below. Where appropriate, analyses involved additional comparison to whole-lake average concentrations of the main basin of Snap Lake and discussion of the likely cause of the elevated (or low, in the case of pH and DO) values. Concentrations in Northeast Lake and Lake 13 are also presented in Table 3-10 for reference.

Table 3-10 Comparison of 2013 Snap Lake Water Quality to AEMP Benchmarks

Parameter	Units	AEMP Benchmarks and SSWQOs (Protection of Aquatic Life)(a)	Observed Concentrations(b)			
			Type	Snap Lake	Northeast Lake	Lake 13
Field Parameters						
Dissolved oxygen	mg/L	6.5 and 9.5(c)	range	**0.3** to 18	**2.1** to 19	**3.2** to 18
pH	unitless	6.5 to 9.0	range	**5.3** to 8.1	**6.1** to 7.3	**6.4** to 7.5
Conventional Parameters						
Laboratory pH	unitless	6.5 to 9.0	range	6.8 to 7.8	**6.4** to 7.3	6.6 to 7.1
Major Ions						
Chloride	mg/L	120 (218 to 388)(d)	max and range in whole-lake average for Snap Lake(e)	**134** and 104 to **128**	1.3	<0.5
Fluoride	mg/L	0.12 (2.46)(f)	max and range in whole-lake average for Snap Lake(e)	**0.23** and **0.13** to **0.18**	0.09	0.06
Nutrients						
Nitrate, as N	mg-N/L	2.93 (4.1 to 16.4)(g)	max and range in whole-lake average for Snap Lake(e)	**3.04** and 2.05 to 2.67	0.018	0.011
Nitrite, as N	mg-N/L	0.06	max	0.027	<0.002	<0.002
Ammonia, as N	mg-N/L	0.41 to 125(h)	max	0.3	0.02	0.01
Total phosphorus(i)	mg-P/L	0.01(j)	max and range in whole-lake average for each lake(e)	0.013 and <0.001 to 0.005	0.023 and 0.002 to 0.003	0.054 and 0.004 to **0.016**
Total Metals						
Aluminum	µg/L	5 to 100(k)	max	8	6	7
Arsenic	µg/L	5	max	0.1	0.1	0.2
Boron	µg/L	1,500	max	69	6	6
Cadmium	µg/L	0.36	max	0.016	<0.005	<0.005
Chromium	µg/L	8.9	max	0.18	0.08	<0.06
Hexavalent chromium	µg/L	2.1	max	1.3	<1	<1
Copper	µg/L	2.1 to 8.1(l)	max	1.1	0.8	1
Iron	µg/L	300	max	12	10	13
Lead	µg/L	1 to 6.96(m)	max	0.03	0.01	<0.01
Mercury (Flett)	µg/L	0.026	max	0.002	0.001	<0.001
Molybdenum	µg/L	73	max	1.73	0.06	<0.05

Table 3-10 Comparison of 2013 Snap Lake Water Quality to AEMP Benchmarks

Parameter	Units	AEMP Benchmarks and SSWQOs (Protection of Aquatic Life)[a]	Observed Concentrations[b]			
			Type	Snap Lake	Northeast Lake	Lake 13
Total Metals (Continued)						
Nickel	µg/L	45.3 to 153[m]	max	2.64	0.35	0.28
Selenium	µg/L	1	max	0.063	<0.04	<0.04
Silver	µg/L	0.1	max	0.006	<0.005	<0.005
Thallium	µg/L	0.8	max	<0.01	<0.01	<0.01
Uranium	µg/L	15	max	0.29	0.01	0.03
Zinc	µg/L	30	max	3	2	1

a) AEMP Benchmarks are: Water Quality Guidelines (WQGs) from the Canadian Council of Ministers of the Environment (CCME) (1999) and site-specific EAR benchmarks developed for the protection of aquatic life for copper, chromium (VI) and cadmium (5% Probable Effect Level) from De Beers (2002). Recommended SSWQOs are included in parentheses when concentrations in Snap Lake exceeded generic AEMP benchmarks.

b) Observed concentrations from the 2013 reporting period (November 1, 2013 to October 31, 2013). Bold values were above an AEMP benchmark.

c) Lowest acceptable dissolved oxygen concentration for cold-water biota is 9.5 mg/L for early life stages, 6.5 mg/L for other life stages.

d) The chloride SSWQO was developed as part of the TDS Response Plan (De Beers 2013h). The SSWQO range was based on the minimum hardness of 37 mg/L observed in Snap Lake during the 2013 reporting period and a maximum hardness of 160 mg/L. Although maximum hardness in Snap Lake was 185 mg/L in 2013, a hardness relationship with toxicity for chloride was not applicable for hardness values above 160 mg/L (Elphick et al. 2011).

e) Range in whole-lake average = minimum and maximum whole-lake average concentrations in Snap Lake, which excludes northwest arm stations.

f) The fluoride SSWQO was developed as part of the TDS Response Plan (De Beers 2013h,i).

g) The nitrate SSWQO was developed as part of Nitrate Response Plan (De Beers 2013j). The SSWQO range was based on the minimum hardness 37 mg/L observed in Snap Lake during the 2013 reporting period and a maximum hardness of 160 mg/L. Although maximum hardness in Snap Lake was 185 mg/L, a hardness relationship with toxicity for nitrate was not applicable for hardness values above 160 mg/L (Rescan 2012).

h) The ammonia WQG is pH and water temperature dependent. The CCME recommended that guideline values falling into the range less than 5 degrees Celsius (°C) and greater than pH of 10 should be used with caution because the lack of toxicity data to accurately determine the toxicity effects at high and low extremes. Therefore, the range of the guideline shown is based on a range of the maximum field pH (8.1) and temperature (16.9°C) in Snap Lake over the 2013 reporting period and the lowest pH (6.0) and temperature (5°C) recommended in (CCME 1999). The guideline was calculated based on an individual pH and water temperature for each sample with the final value expressed as ammonia nitrogen.

i) Total phosphorus (TP) concentrations were presented as both maximum observed concentrations and ranges in whole-lake averages collected using the water quality and plankton component methods. Total phosphorus whole-lake averages were compared to the AEMP benchmark.

j) The TP benchmark was derived from Wetzel (2001); mesotrophic conditions were defined by TP of 10.9 to 95.6 µg/L. The benchmark was based on the lower end of this range.

k) The aluminum WQG is pH dependent. The guideline shown is based on a range of field pH observed in Snap Lake during the 2013 reporting period (5.3 to 8.1). The WQG was calculated based on the individual pH for each sample.

l) The copper site-specific EAR benchmark was based on the minimum hardness of 37 mg/L and maximum of 185 mg/L which were observed in Snap Lake during the 2013 reporting period.

m) The lead and nickel WQGs are hardness dependent. The range of the WQGs shown was based on a range of hardness observed in Snap Lake during the 2013 reporting period (37 to 185 mg/L). The WQG was calculated based on the individual hardness for each sample.

N = nitrogen; - = not applicable; < = less than; min = minimum; max = maximum; Flett = Flett Research Limited; µg/L = micrograms per litre; mg/L = milligrams per litre; mg-N/L = milligrams as nitrogen per litre; AEMP = Aquatic Effects Monitoring Program.

Chloride

In 2013, approximately 21% of measured chloride concentrations in Snap Lake were above the CCME WQG of 120 mg/L, with the maximum concentration of 134 mg/L being measured at station SNP 02-20d. Elevated chloride concentrations in Snap Lake are due to the discharge of treated effluent, which contains elevated concentrations of chloride from groundwater sources. Whole-lake average chloride concentrations ranged from 104 mg/L to 128 mg/L and the maximum whole-lake average concentration, which occurred in May, was also above the CCME WQG (Table 3-10).

The measured chloride concentrations are not expected to cause adverse effects to aquatic biota in Snap Lake. The toxicity of chloride decreases with increases in hardness (CCME 2011; Gills 2011; Elphick et al. 2011), which occurs in Snap Lake concurrently with chloride increases. This is also due to discharges of treated effluent. The recommended SSWQO was developed for chloride in Snap Lake as part of the TDS Response Plan using a hardness-based formula (Elphick et al. 2011, De Beers 2013h). The maximum 2013 chloride concentration in Snap Lake (134 mg/L) was below the lowest chloride SSWQO (218 mg/L; calculated using the minimum observed hardness of 37 mg/L).

Nitrate

Approximately 3% of the 2013 nitrate samples collected in Snap Lake were above the CCME WQG for nitrate of 2.93 milligrams as nitrogen per litre (mg-N/L), with the maximum concentration of 3.04 mg-N/L measured at station SNP 02-20f. Whole-lake average nitrate concentrations ranged from 2.1 mg/L to 2.7 mg/L, and remained below the CCME WQG in 2013 (Table 3-10). Elevated nitrate concentrations in Snap Lake are due to discharge of treated effluent, which contains nitrogen compounds from treated domestic waste water and explosives used in the Mine.

The observed nitrate concentrations are not expected to cause adverse effects to aquatic biota in Snap Lake. The toxicity of nitrate decreases with increases in hardness (Rescan 2012; De Beers 2013j). The recommended SSWQO for nitrate was developed as part of the Nitrate Response Plan using a hardness-based formula (Rescan 2012; De Beers 2013j). The maximum 2013 nitrate concentration in Snap Lake (3.04 mg-N/L) was below the lowest nitrate SSWQO of 4.1 mg/L.

Fluoride

More than half of the samples (i.e., 63%) collected in 2013 were higher than the 2001 interim CCME (2002) WQG for inorganic fluoride of 0.12 mg/L. The maximum fluoride concentration measured in 2013 was 0.23 mg/L at station SNP 02-20f (Table 3-10). Elevated fluoride concentrations in Snap Lake are due to the discharge of treated effluent, which contains fluoride from groundwater sources.

The observed fluoride concentrations are not expected to cause adverse effects to aquatic biota in Snap Lake. The toxicity of fluoride is expected to decrease with increases in hardness, chloride, and calcium (Environment Canada 2001, CCME 2002). Fluoride increases in Snap Lake will occur concurrently with hardness, chloride and calcium increases in Snap Lake because the treated effluent contains elevated concentrations of all four parameters. The recommended SSWQO for fluoride was developed as part of the TDS Response Plan using available acute and chronic toxicity studies for fluoride, which tested freshwater aquatic life species that are relevant to Snap Lake (De Beers 2013h,i). The maximum 2013 fluoride concentration in Snap Lake (0.23 mg/L) was below the fluoride SSWQO of 2.46 mg/L.

pH

Field pH values were generally within the CCME WQG pH range of 6.5 to 9.0 (CCME 1999) in 2013 (Appendix 3C) with some exceptions. Field pH values were consistently low and below the CCME pH range in the main basin and the northwest arm of Snap Lake in March 2013 (Section 3.4.5). Values of field pH were also below the CCME WQG pH range in Northeast Lake in 2013 (Table 3-10). Field pH values have occasionally been below the CCME range in previous years during Mine operations and during ice-covered baseline conditions (De Beers 2002), but the overall temporal trend in both laboratory and field pH values is increasing (Section 3.4.4). Both laboratory and field pH values in 2013 were above the normal range and the reference lakes ranges (i.e., baseline mean ± two SDs and reference lakes mean ± two SDs, respectively; Section 3.4.4). Since historical field pH values, including baseline values, and reference lakes values were below the CCME guideline; range and pH values are generally increasing in Snap Lake, the occasional low field pH values in Snap Lake were more likely due to natural variations in pH rather than Mine-related causes.

Dissolved Oxygen

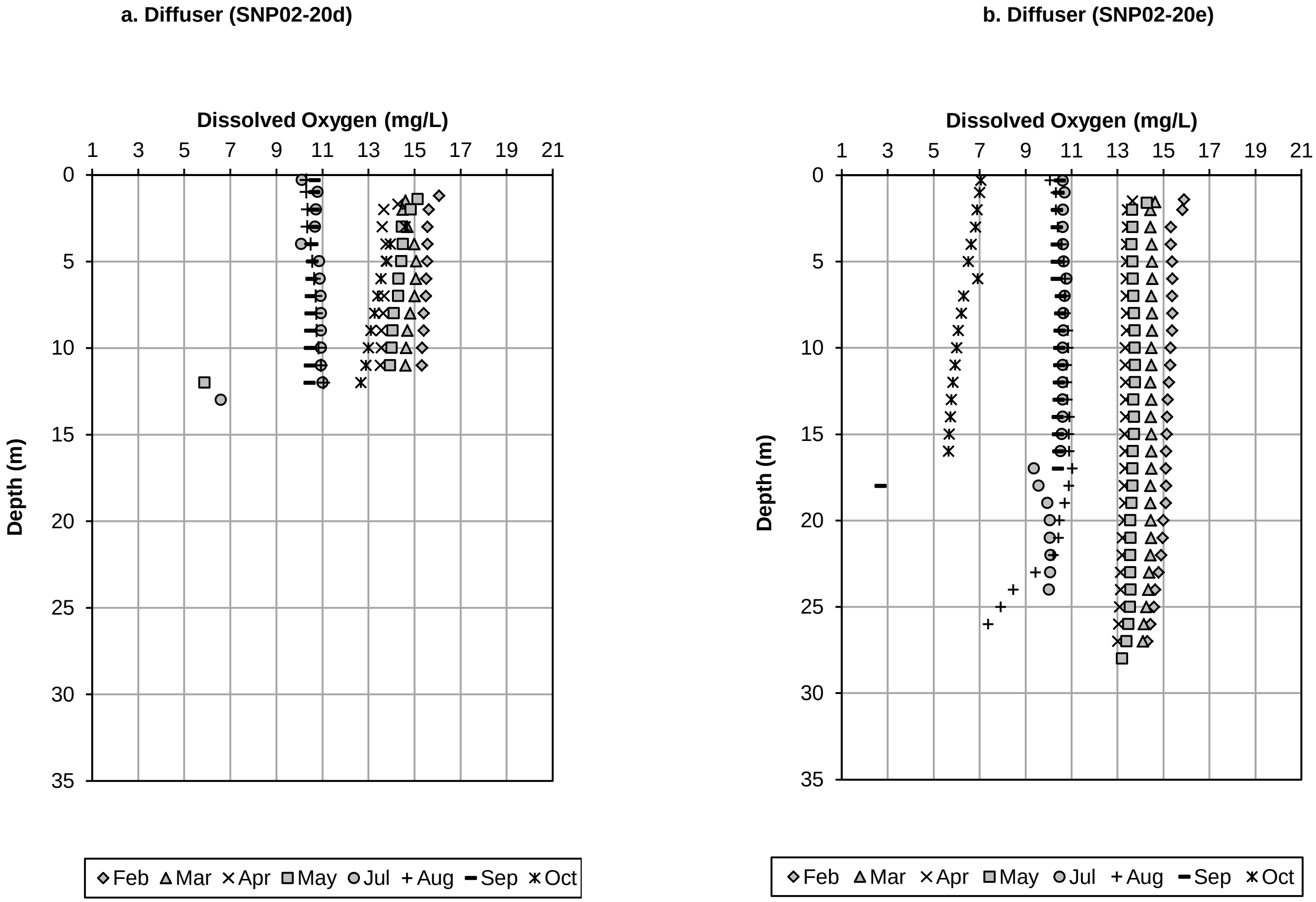
The DO concentration in lake water is a function of the balance of the processes that introduce or supplement oxygen into the water column, and remove oxygen from the water column. Re-aeration by the diffuser and wind-mixing as well as photosynthesis by algae and aquatic plants are prime examples of processes that introduce oxygen into the water column. Respiration by algae and aquatic biota, microbial decomposition of organic matter in the water column and at the surface of bottom sediments, and the oxidation or nitrification of ammonia are examples of processes that consume or remove oxygen from the water column.

In 2013, DO concentrations in Snap Lake were considered healthy for fish and other aquatic organisms, with the exception of six locations (Figure 3-7) where field DO readings dropped below the CCME WQG of 6.5 mg/L (CCME 1999). At three of these locations (SNAP29, SNAP02A, and SNP 02-20d), the low DO only occurred in the bottom 0.5 m of the water column, indicating that the measurement probe was likely near the sediment boundary or possibly submerged in sediment as denoted by substantial changes in DO. As outlined in Appendix 3A, DO data from near the bottom of the water column were only excluded from the assessment if there was a large corresponding change in pH, temperature, and/or conductivity, which could indicate a submerged probe measuring sediment pore water quality. At the other three stations (SNAP20B, SNAP23, and SNP 02-20e), which are the deepest stations in Snap Lake, the DO decreased starting from either the surface or mid-depth to the bottom of the lake. The lack of re-aeration potential due to ice-cover and oxygen consumption through natural biological and chemical processes in the water column could cause naturally low bottom DO concentrations in lakes during winter conditions

(Catalan et al. 2002). The DO concentration at one station in Northeast Lake (NEL06) and one station in Lake 13 (LK13-1) also decreased from the mid-depth to the bottom of the lake, and the DO readings dropped below the CCME WQG of 6.5 mg/L (Figure 3-7).

Since routine DO profile monitoring began in 2007, low DO concentrations near the bottom of the lake have been measured (De Beers 2002). Overall, DO concentrations in Snap Lake did not decrease as a result of treated effluent discharge. Increases, rather than decreases of DO concentrations, have occurred over time as presented in Section 3.4.4.

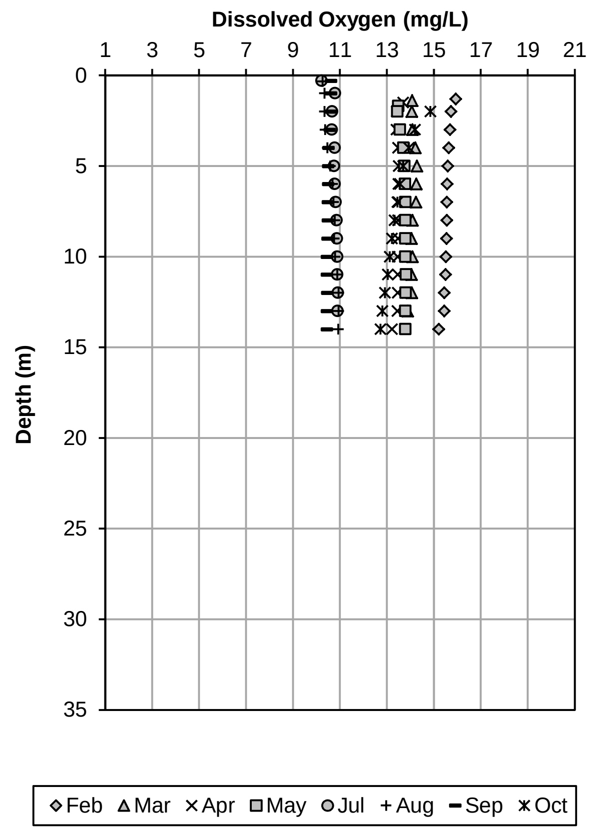
Figure 3-7 Dissolved Oxygen Concentrations in Snap Lake, Northeast Lake and Lake 13, 2013



m = metre; mg/L = milligrams per litre.

Figure 3-7 Dissolved Oxygen Concentrations in Snap Lake, Northeast Lake, and Lake 13, 2013

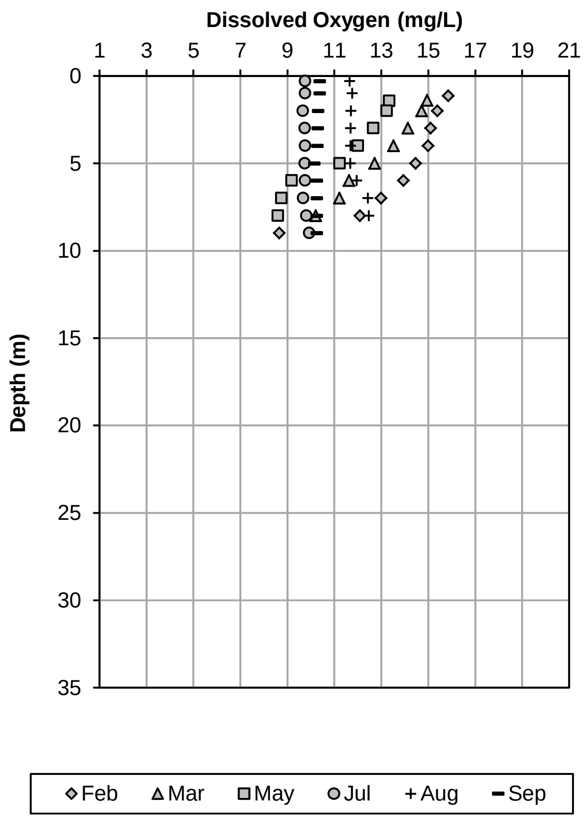
c. Diffuser (SNP02-20f)



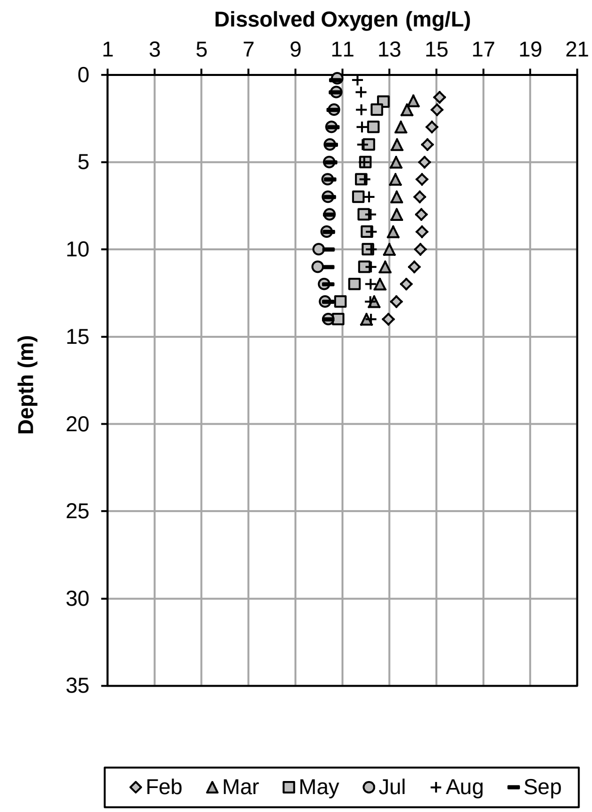
m = metre; mg/L = milligrams per litre.

Figure 3-7 Dissolved Oxygen Concentrations in Snap Lake, Northeast Lake and Lake 13, 2013

d. Main Basin (SNAP08)



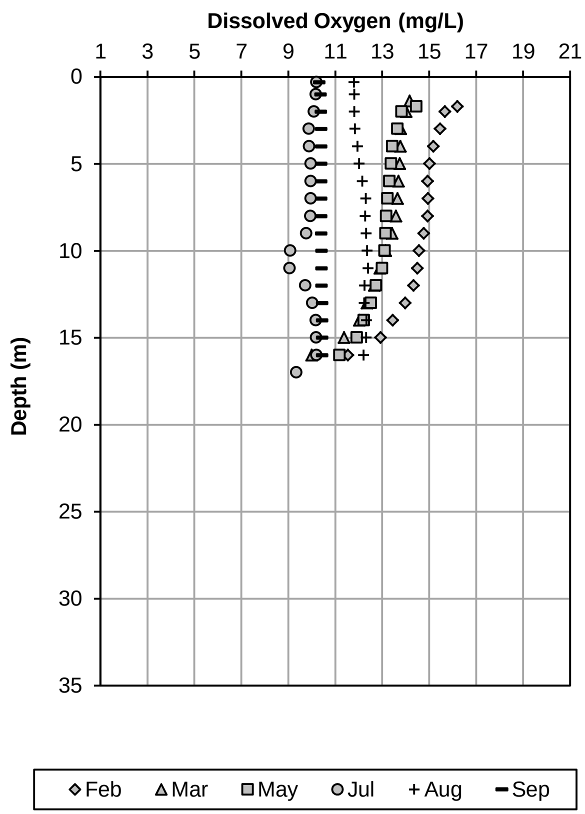
e. Main Basin (SNAP09)



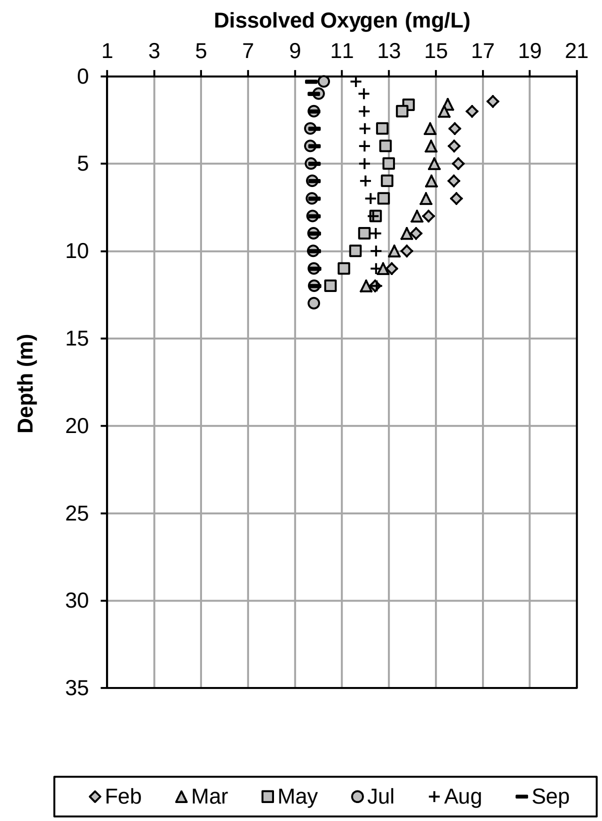
m = metre; mg/L = milligrams per litre.

Figure 3-7 Dissolved Oxygen Concentrations in Snap Lake, Northeast Lake and Lake 13, 2013

f. Main Basin (SNAP11A)



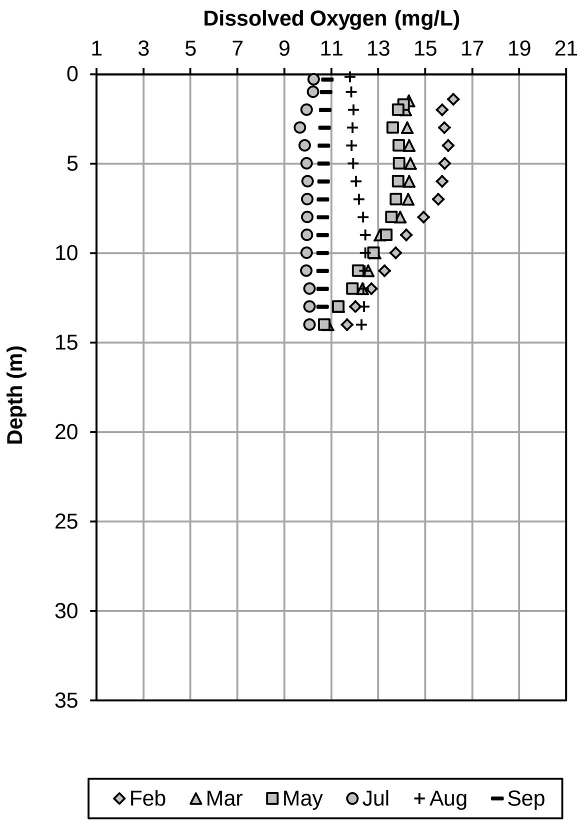
g. Main Basin (SNAP06)



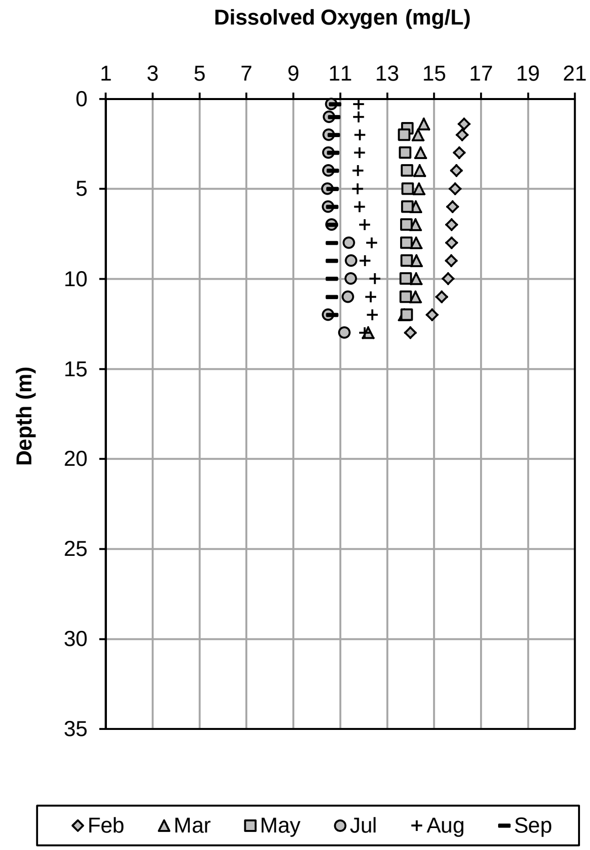
m = metre; mg/L = milligrams per litre.

Figure 3-7 Dissolved Oxygen Concentrations in Snap Lake, Northeast Lake and Lake 13, 2013

h. Main Basin (SNAP05)



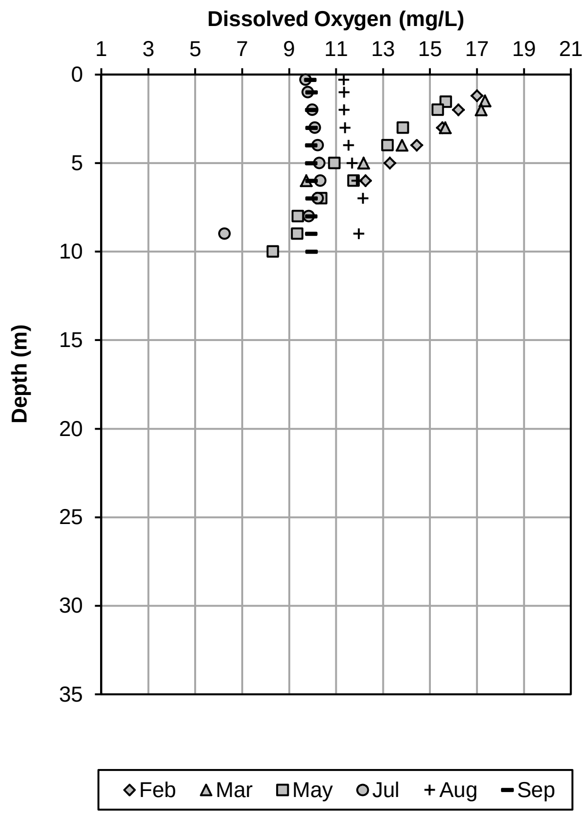
i. Main Basin (SNAP03)



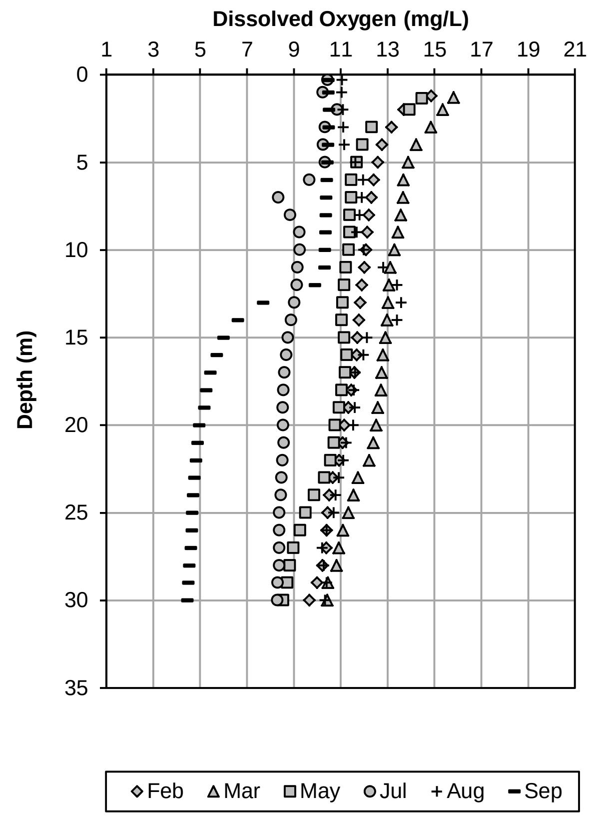
m = metre; mg/L = milligrams per litre.

Figure 3-7 Dissolved Oxygen Concentrations in Snap Lake, Northeast Lake and Lake 13, 2013

j. Northwest Arm (SNAP02A)



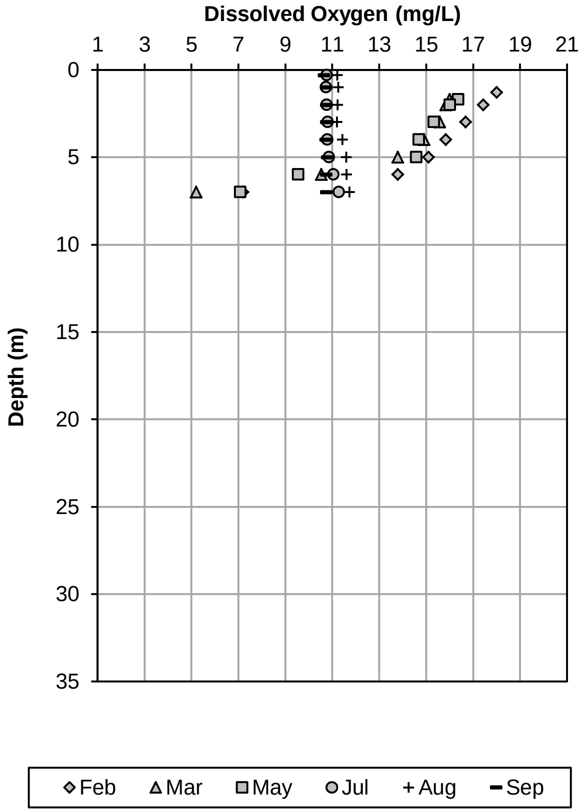
k. Northwest Arm (SNAP20B)



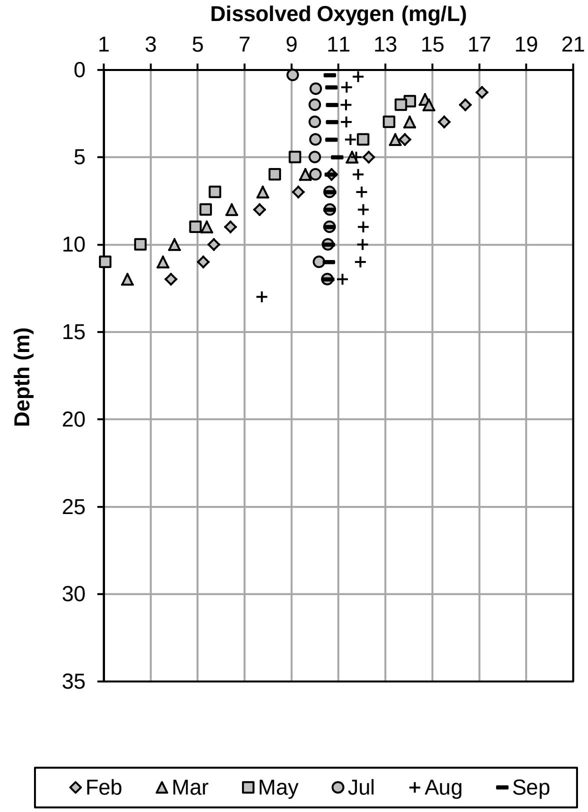
m = metre; mg/L = milligrams per litre.

Figure 3-7 Dissolved Oxygen Concentrations in Snap Lake, Northeast Lake and Lake 13, 2013

l. Northwest Arm (SNAP29)

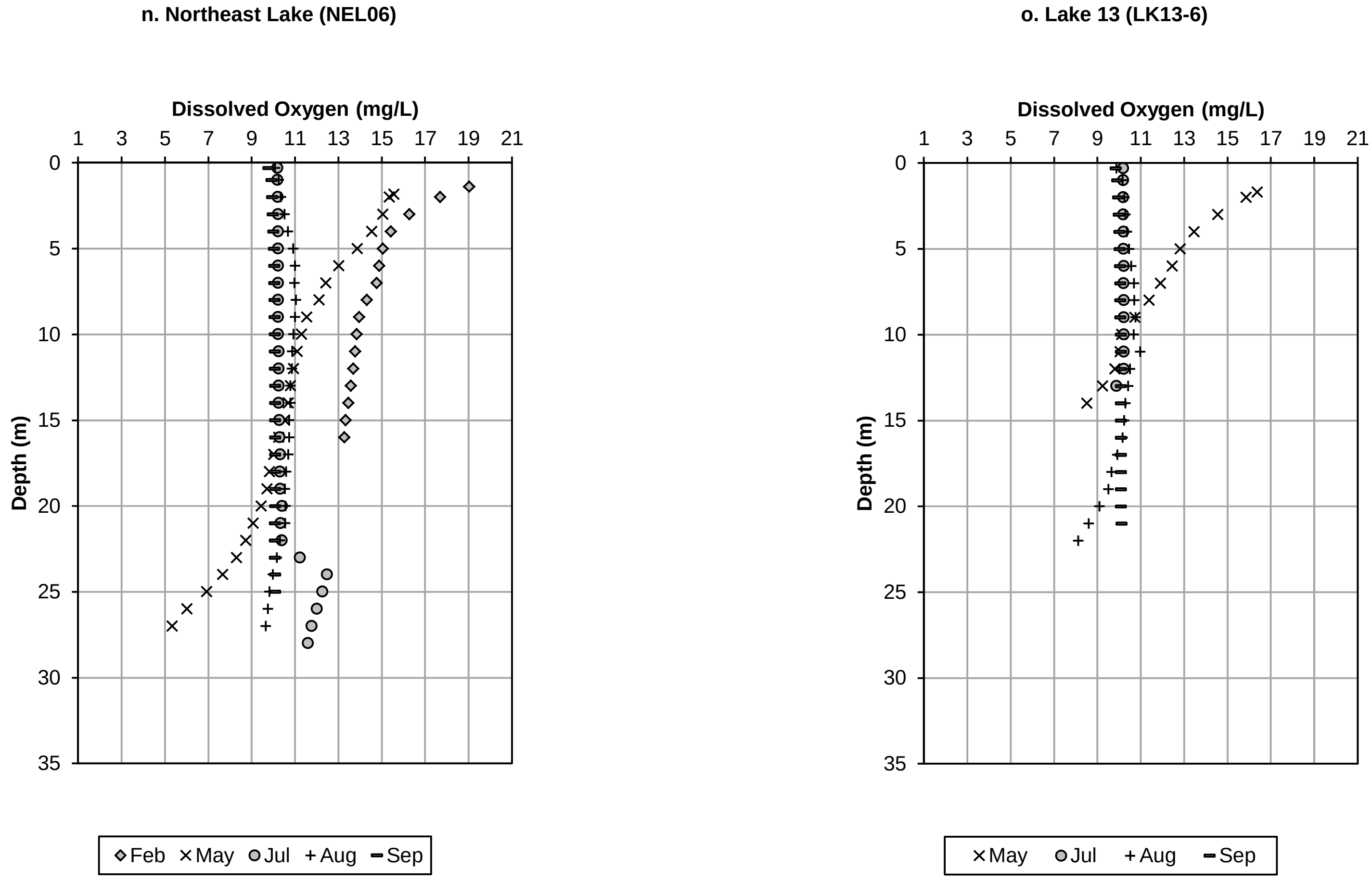


m. Northwest Arm (SNAP23)



m = metre; mg/L = milligrams per litre

Figure 3-7 Dissolved Oxygen Concentrations in Snap Lake, Northeast Lake and Lake 13, 2013



m = metre; mg/L = milligrams per litre.

3.4.3.2 AEMP Benchmark Comparison for Action Level Assessment

Monthly average concentrations of parameters measured at the edge of mixing zone (i.e., at the diffuser stations) were compared to 75% of the AEMP Benchmarks to assess whether a low Action Level for toxicological impairment would be triggered (Table 3-11). For nutrient enrichment, the maximum whole-lake average total phosphorus concentration was compared to 75% of the AEMP nutrient benchmark.

Based on these comparisons, relevant average concentrations were below 75% of the AEMP Benchmarks with the exception of chloride, fluoride, and nitrate. For a toxicological impairment Action Level to be triggered, parameter concentrations must be greater than 75% of the AEMP benchmark *and* concentrations must be greater than normal and reference ranges supported by an increasing trend (Table 3-7). Trends in chloride, fluoride, and nitrate were evaluated and compared to reference lakes in Key Question 3 (Section 3.4.4). The maximum whole-lake average for total phosphorus was below 75% of the nutrient benchmark, thus no Action Levels related to nutrient enrichment were triggered. No Action Levels related to toxicity testing were triggered.

The results of the water quality Action Level assessment (i.e., benchmark comparisons, trend assessment, and comparison to normal and reference ranges) are integrated in Section 3.4.8 and a relevant summary of triggered action levels are provided in Section 13.

Table 3-11 Comparison of 2013 Snap Lake Monthly Average Water Quality Concentrations to AEMP Benchmarks

Parameter	Unit	AEMP Benchmarks[(a)]	Maximum Monthly Average	
			Value[(b)]	Percent of AEMP Benchmark
Major Ions				
Chloride	mg/L	120	**133**	>100%
Fluoride	mg/L	0.12	**0.2**	>100%
Nutrients				
Nitrate, as N	mg-N/L	2.93	**2.98**	>100%
Nitrite, as N	mg-N/L	0.06	0.023	38%
Total Ammonia, as N	mg-N/L	2.69[(c)]	0.29	11%
Total Phosphorus	mg-P/L	0.011	0.005	45%
Total Metals				
Aluminum	µg/L	100[(d)]	7	7%
Arsenic	µg/L	5	0.1	2%
Boron	µg/L	1,500	68	5%
Cadmium	µg/L	0.36	0.005	1%
Chromium	µg/L	8.9	0.08	1%
Hexavalent Chromium	µg/L	2.1	<1	<1%

Table 3-11 Comparison of 2013 Snap Lake Monthly Average Water Quality Concentrations to AEMP Benchmarks

Parameter	Unit	AEMP Benchmarks[a]	Maximum Monthly Average	
			Value[b]	Percent of AEMP Benchmark
Total Metals (Continued)				
Copper	µg/L	3.1[e]	0.5	16%
Iron	µg/L	300	8	3%
Lead	µg/L	6.3[e]	0.02	<1%
Mercury (Flett)	µg/L	0.026	<0.0005	<1%
Molybdenum	µg/L	73	1.68	2%
Nickel	µg/L	149[e]	2.62	2%
Selenium	µg/L	1	0.05	5%
Silver	µg/L	0.1	<0.005	<1%
Thallium	µg/L	0.8	<0.01	<1%
Uranium	µg/L	15	0.29	2%
Zinc	µg/L	30	2	7%

Note: Bolded concentrations are higher than 75% of the relevant AEMP benchmark.

a) AEMP benchmarks are: Water Quality Guidelines (WQGs) from the Canadian Council of Ministers of the Environment (CCME) (1999) and site-specific Environment Assessment Report (EAR) benchmarks from De Beers (2002). Additional detail on benchmarks is provided in Table 3-10.

b) Maximum monthly averages for each parameter were calculated using the diffuser stations results from February to October 2013, with the exception of total phosphorus (TP). For TP, the maximum whole-lake average was compared to 75% of the AEMP nutrient benchmark. Concentrations reported as below detection limit were set to half the detection limit when calculating average concentrations.

c) The ammonia WQG is based on the monthly average of laboratory pH (7.9) and temperature (1.4°C) in May 2013, when the maximum monthly average total ammonia concentration was observed. Field pH was replaced with laboratory pH based as per the quality assurance and quality control (QA/QC) assessment (Appendix 3A).

d) The aluminum WQG is based on the monthly average field pH of 7.8 in August 2013, when the maximum aluminum average concentration was observed.

e) The copper site-specific EAR benchmark and WQGs for nickel and lead guidelines were based on a hardness of 139 mg/L, 180 mg/L, and 171 mg/L in August, May, and April in 2013, respectively, when the maximum monthly average concentrations were observed.

< = less than; > = greater than; N = nitrogen; P = phosphorus; Flett = Flett Research Limited; mg/L = milligram per litre; µg/L=micrograms per litre; mg-N/L = milligram as nitrogen per litre; mg-P/L = milligram per as phosphorus per litre; % = percent.

3.4.3.3 Whole-Lake Average Water Licence Limit for Total Dissolved Solids

Whole-lake average TDS concentrations in 2013 ranged from 228 to 284 mg/L and were below the Water Licence limit of 350 mg/L for the whole-lake average (Table 3-12). Although whole-lake averages remained below 350 mg/L in 2013, TDS is predicted to exceed 350 mg/L between January 2014 and January 2015, depending on the scenario modelled in the water quality modelling update (De Beers 2013g). Therefore, the TDS Response Plan, which included detailed discussion of TDS sources, management, and effects on aquatic life recommended, with supporting rationale, a SSWQO for TDS of 684 mg/L (De Beers 2013h). Concentrations of TDS in 2013 remained well below the TDS SSWQO.

Table 3-12 Comparison of Whole-Lake Average to Water Licence Limit for Total Dissolved Solids, 2013

Water Licence Limit (mg/L)[a]	Whole-Lake Average Concentration for Total Dissolved Solids (mg/L)				
	Ice-Covered		Open-Water		
	February	April	July	August	September
350	254	284	228	232	242

a) Water Licence limit issued in MV2011L2-0004 (MVLWB 2013a).
mg/L = milligram per litre.

3.4.3.4 Toxicity Test Results for Snap Lake

Toxicity test results provide information about toxicity to representative aquatic organisms, including potential for sublethal effects. Collectively, the results from these tests are used to determine whether there is a spatial or temporal trend in toxicity that needs to be investigated further. Three diffuser samples were collected and tested for toxicity in May and three in September. There were no adverse effects for any test endpoints (i.e., no toxicity in Snap Lake). Algal growth was stimulated in all samples, with the degree of stimulation increasing at higher sample concentrations (Appendix 3F).

The results of toxicity testing did not trigger any Action Levels because persistent chronic toxicity was not observed in the lake samples.

3.4.3.5 Summary of Key Question 2

The 2013 water quality data from Snap Lake were below AEMP benchmarks and Water Licence limits, with the exception of chloride, fluoride, and nitrate. Because the primary source of parameters is the treated effluent, increases in these parameters are associated with elevated calcium and hardness, which reduce the potential for toxicity effects associated with fluoride, chloride, and nitrate. Chloride concentrations in Snap Lake were often above the CCME WQG of 120 mg/L for chloride, but the maximum chloride concentration was below the lowest SSWQO recommended for chloride. Approximately 3% of the 2013 nitrate samples collected in Snap Lake were above the CCME WQG for nitrate of 2.93 mg-N/L, with a maximum concentration of 3.04 mg-N/L. The nitrate whole-lake average concentrations were below the CCME WQG, and maximum concentrations were below the lowest SSWQO recommended for nitrate. The majority of the 2013 fluoride concentrations were above the interim CCME WQG of 0.12 mg/L. The maximum 2013 fluoride concentration in Snap Lake was below the recommended fluoride SSWQO of 2.46 mg/L.

Low field pH values were measured in Snap Lake and the reference lake, Northeast Lake, in 2013; values were occasionally below the CCME WQG of 6.5. Low field pH values have also been measured in Snap Lake in previous years during Mine operations and under baseline conditions during the ice-covered season. Both field pH and laboratory pH values have increased over time in Snap Lake to values above the normal and the reference lakes range (Section 3.4.4). Therefore, the occasional low pH value observed in 2013 was likely due to natural variation in pH values because the overall trend in pH in Snap Lake was increasing.

In 2013, DO concentrations in Snap Lake were considered healthy for fish and other aquatic organisms, with the exception of six locations, where field DO readings dropped below the CCME WQG of 6.5 mg/L. At three of these locations, the low DO was limited to the bottom 0.5 m of the water column. Low DO concentrations near the bottom of the lake were observed during ice-covered baseline conditions. The DO concentrations decreased from the mid-depth to the bottom at the deepest stations in the Snap Lake; low DO concentrations near the bottom of Snap Lake have been measured since 2007. The DO concentration at one station in Northeast Lake, and one station in Lake 13 also decreased from the mid-depth to the bottom of the lake, and dropped below the CCME WQG of 6.5 mg/L. The lack of re-aeration potential due to ice-cover and oxygen consumption through natural biological and chemical processes in the water column can cause naturally low bottom DO concentrations in lakes during winter conditions (Catalan et al. 2002).

Whole-lake average TDS concentrations ranged from 228 mg/L to 284 mg/L, with a maximum TDS concentration of 301 mg/L, all below the Water Licence limit of 350 mg/L. Maximum monthly average concentrations at the diffuser stations were below values which would trigger Action Levels related to toxicological impairment (i.e., 75% of the AEMP Benchmarks), with the exception of chloride, fluoride, and nitrate. To determine whether Action Levels were triggered for chloride, fluoride, and nitrate, tests for temporal trends and comparisons to reference lake concentrations were completed for these three parameters in Key Question 3 (Section 3.4.4). The combined results from Key Questions 2 and 3 were used to determine whether Action Levels were triggered (Section 3.4.8). Values of pH and DO concentrations were occasionally below AEMP benchmarks but the low values were determined not to be Mine-related. The maximum whole-lake average for total phosphorus was below 75% of the nutrient benchmark so no Action Levels related to nutrient enrichment were triggered. No Action Levels related to toxicity testing were triggered. The results of the water quality Action Level assessment are integrated in Section 3.4.8.

3.4.4 Key Question 3: Which Water Quality Parameters are Increasing over Time in Snap Lake, and How do Concentrations of these Parameters compare to AEMP Benchmarks, Concentrations in Reference Lakes, EAR Predictions, and Subsequent Modelling Predictions?

The EAR predicted that major ions, nutrients, and some metals would increase, and DO would decrease, in Snap Lake due to the discharges of treated effluent from the Mine. The results of the 2013 temporal assessment are described in the following subsections.

3.4.4.1 Screening Based on Visual Evaluation of Temporal Plots

Parameters that were Strongly Correlated with Conductivity

Correlation analysis identified 44 water quality parameters with concentrations significantly related to conductivity in Snap Lake between 2004 and 2013 (Table 3-13). Parameters that were strongly correlated to conductivity were considered chemical signatures of treated effluent influence. Within this group of parameters, 23 increased in both the main basin and the northwest arm, and two increased in only the main basin. Increasing concentrations of TDS, most major ions, most nitrogen parameters, and seven metals (barium, boron, lithium, molybdenum, nickel, rubidium, and strontium) were observed throughout Snap Lake, including the northwest arm. Fluoride concentrations are increasing in the main basin of Snap Lake and during ice-covered conditions in the northwest arm; increases in fluoride in the northwest arm during open-water conditions were not observed (Table 3-13). Reactive silica and total uranium showed increasing trends in the main basin of Snap Lake, but not in the northwest arm (Table 3-13).

An increasing trend of nitrite in the northwest arm was new in 2013. The greater number of water quality parameters with increasing trends identified in the northwest arm indicates that the treated effluent is gradually changing the water quality within the northwest arm.

Examples of temporal trends are shown for stations representative of areas within Snap Lake (i.e., diffuser, main basin, and northwest arm) and the reference lakes (i.e. Northeast Lake and Lake 13) for TDS, TN, and total strontium for the parameters that strongly correlated with conductivity and demonstrated an increasing trend within one or more lake areas (Figures 3-8 to 3-10, respectively).

The visual review of temporal trends in TN included results from both the water quality and plankton components; the water quality and plankton phosphorus trends were reviewed separately because they were collected using different methods. Distinct increasing trends in both water quality and plankton TN were observed throughout Snap Lake (Figure 3-9).

Parameters that were Moderately Correlated with Conductivity

Five parameters were moderately correlated with conductivity; two of these parameters had positive correlations (i.e., laboratory pH and total Kjeldahl nitrogen [TKN]) and three parameters had negative correlations (i.e., total aluminum, mercury, and thallium) (Table 3-13). Laboratory pH and TKN were increasing in Snap Lake over time (Table 3-13).

Parameters that are Weakly Correlated with Conductivity

Fourteen parameters had weak (or low) correlations with conductivity; two of these had positive correlations (i.e., field pH and titanium), and 12 parameters had negative correlations (i.e., turbidity, dissolved inorganic phosphorus, dissolved organic phosphorus, dissolved phosphorus, total inorganic phosphorus, total organic carbon, TP; and total cobalt, copper, iron, lead, and mercury; Table 3-13). Temporal trends were not observed for any of the 14 weakly correlated parameters in the main basin or northwest arm of Snap Lake, with the exception of field pH. Based on a visual review, field pH may be increasing over time in Snap Lake; however, statistical trend analysis completed to determine whether the trend was statistically significant (Seasonal Kendall Test in Section 3.4.4.2).

The visual review of temporal trends in TP collected as part of the water quality and plankton components indicated that phosphorus is not increasing in Snap Lake (Figure 3-11). Similar to TN, water quality and plankton phosphorus trends were reviewed separately because data were collected using different methods. Although phosphorus is in the treated effluent, primarily due to treated domestic waste water, increases in this parameter may not be obvious due to the following possible reasons:

- a small increase in phosphorus may not be detected due to analytical uncertainties at the low phosphorus concentrations in Snap Lake;
- phosphorus may act non-conservatively, due to natural biological uptake or sedimentation processes; or,
- a combination of analytical uncertainties and these natural processes.

The analytical uncertainties in Snap Lake phosphorus concentrations have been estimated to be approximately plus or minus (±) 0.002 mg/L (Appendix 3B), so an increase in this range may not be detectable. Plankton in the water column that take up phosphorus and subsequently die and settle to the bottom of the lake may be removing phosphorus from the water column. Increases in sediment phosphorus were not observed, and trends identified for available phosphate at the diffuser stations were inconsistent (i.e., increased from 2007 to 2011, then decreased in 2012 and 2013; Section 4.4.4); however, increases in sediment phosphorus may not have been detected if increases were small enough relative to the natural variability in sediment phosphorus. Statistical trend analysis was completed for TP to confirm the lack of apparent temporal trends for this parameter (Seasonal Kendall Test in 3.4.4.2).

Parameters with No Correlations with Conductivity

Ortho-phosphate, total antimony, cadmium, chromium, manganese, silver, and zinc had no correlation to conductivity, and therefore were not reviewed for trends, with the exception of manganese. In the spatial analysis results, Snap Lake concentrations of manganese were greater than the reference lakes (Figure 3-12, Section 3.4.5.1). Manganese, which is in the treated effluent at concentrations similar to barium, could be increasing in Snap Lake without correlating with conductivity due to natural reduction and oxidation processes that are controlled primarily by DO concentrations. Oxidation of manganese results in this metal becoming insoluble and potentially settling out of the water column; corresponding temporal trends in sediment manganese have not been observed in the main basin of Snap Lake (Section 4.4.4), but increases in sediment manganese may not be detectable if they are small relative to natural variability. The water quality temporal trend plots indicate that manganese may be increasing in Snap Lake, most noticeably in surface concentrations at the diffuser (Figure 3-12). Statistical trend analysis was completed for manganese to confirm the apparent increases in this parameter observed in the Snap Lake temporal plots (Seasonal Kendall Test in Section 3.4.4.2).

Comparisons to Reference Lake and Baseline Normal Range

Temporal trends were not observed in Northeast Lake and Lake 13 between 2004 and 2013 for any of the measured parameters. Concentrations of most parameters with increasing trends observed in one or more areas of Snap Lake were above the Snap Lake normal range (i.e., baseline mean ± two SDs) and above the reference lake range with the exception of TKN (Table 3-13). The TKN concentrations remained within the normal range. The lower concentrations and absence of trends in Northeast Lake and Lake 13 indicate that treated effluent exposure from the Mine is the primary contributor to the observed increases in pH, conductivity, TDS, major ions, nitrogen parameters, and eight metals in Snap Lake.

Additional temporal plots of parameters from Snap Lake and from reference lakes, Northeast Lake and Lake 13 are presented in Appendix 3G, Figures 3G-1 to 3G-51.

Table 3-13 Summaries of Temporal Trends that Were Significantly Correlated with Laboratory Conductivity

Parameter	Pearson Correlation Coefficient	Strength of Correlation Coefficient[a]	Temporal Trends in Snap Lake Area (2004 to 2013)			2013 Maximum Concentration Above Snap Lake Normal Range	2013 Snap Lake Whole-Lake Average Concentration Above Reference Lake Normal Range
			Diffuser	Main Basin	Northwest Arm		
Field Parameters							
pH	0.283	Low	↑	↑	↑	N/A	N/A
Conventional Parameters							
Laboratory pH	0.660	Moderate	↑	↑	↑	yes	yes
Total Dissolved Solids, Calculated	0.998	High	↑	↑	↑	yes	yes
Turbidity	-0.173	Low	-	-	-	no	no
Major Ions							
Bicarbonate, as HCO_3	0.915	High	↑	↑	↑	yes	yes
Calcium	0.994	High	↑	↑	↑	yes	yes
Chloride	0.997	High	↑	↑	↑	yes	yes
Fluoride	0.863	High	↑	↑	-	yes[b]	yes
Hardness, as $CaCO_3$	0.994	High	↑	↑	↑	yes	yes
Magnesium	0.976	High	↑	↑	↑	yes	yes
Potassium	0.949	High	↑	↑	↑	yes	yes
Reactive Silica, as SiO_2	0.760	High	↑	↑	-	yes[c]	yes
Sodium	0.993	High	↑	↑	↑	yes	yes
Sulphate	0.988	High	↑	↑	↑	yes	yes
Total Alkalinity, as $CaCO_3$	0.915	High	↑	↑	↑	yes	yes

Table 3-13 Summaries of Temporal Trends that Were Significantly Correlated with Laboratory Conductivity

Parameter	Pearson Correlation Coefficient	Strength of Correlation Coefficient[a]	Temporal Trends in Snap Lake Area (2004 to 2013)			2013 Maximum Concentration Above Snap Lake Normal Range	2013 Snap Lake Whole-Lake Average Concentration Above Reference Lake Normal Range
			Diffuser	Main Basin	Northwest Arm		
Nutrients and Biological Indicators							
Dissolved Inorganic Phosphorus	-0.191	Low	-	-	-	no	no
Dissolved Organic Phosphorus, Calculated	-0.124	Low	-	-	-	no	no
Dissolved Phosphorus	-0.242	Low	-	-	-	no	no
Nitrate, as N, Calculated	0.941	High	↑	↑	↑	yes	yes
Nitrate/Nitrite, as N	0.938	High	↑	↑	↑	yes	yes
Nitrite, as N	0.851	High	↑	↑	↑	yes	yes
Total Ammonia, as N	0.765	High	↑	↑	↑	yes[c]	yes
Total Inorganic Phosphorus	-0.251	Low	-	-	-	no	no
Total Kjeldahl Nitrogen	0.597	Moderate	↑	↑	-	no	yes
Total Nitrogen	0.934	High	↑	↑	↑	yes[c]	yes
Total Organic Carbon	-0.140	Low	-	-	-	no	no
Total Phosphorus	-0.279	Low	-	-	-	no	no
Total Metals							
Total Aluminum	-0.347	Moderate	-	-	-	no	no
Total Barium	0.965	High	↑	↑	↑	yes	yes
Total Boron	0.957	High	↑	↑	↑	yes	yes
Total Cobalt	-0.116	Low	-	-	-	no	yes[d]
Total Copper	-0.180	Low	-	-	-	no	no
Total Iron	-0.134	Low	-	-	-	no	no
Total Lead	-0.259	Low	-	-	-	no	no
Total Lithium	0.983	High	↑	↑	↑	yes[b]	yes
Total Mercury	-0.185	Low	-	-	-	no	no
Total Mercury (Flett)	-0.340	Moderate	-	-	-	no	no
Total Molybdenum	0.941	High	↑	↑	↑	yes	yes

Table 3-13 Summaries of Temporal Trends that Were Significantly Correlated with Laboratory Conductivity

Parameter	Pearson Correlation Coefficient	Strength of Correlation Coefficient[(a)]	Temporal Trends in Snap Lake Area (2004 to 2013)			2013 Maximum Concentration Above Snap Lake Normal Range	2013 Snap Lake Whole-Lake Average Concentration Above Reference Lake Normal Range
			Diffuser	Main Basin	Northwest Arm		
Total Metals (Continued)							
Total Nickel	0.850	High	↑	↑	↑	yes[(e)]	yes
Total Rubidium	0.915	High	↑	↑	↑	yes	yes
Total Strontium	0.993	High	↑	↑	↑	yes	yes
Total Thallium	-0.390	Moderate	-	-	-	no	no
Total Titanium	0.166	Low	-	-	-	no	no
Total Uranium	0.846	High	↑	↑	-	yes[(c)]	yes

Note: The normal range is based on data collected prior to 2004, with the upper and lower range calculated as the mean concentration ± 2 standard deviations. For parameters which were typically below the detection limits, the detection limit was used as the normal range. The 2013 Snap Lake whole-lake average concentrations were compared to the reference lakes mean concentration from the 2013 monitoring period ± 2 standard deviations.

a) The strength of the correlations was classified as low (r <0.3), moderate (r between 0.4 and 0.7), or high (r >0.7) based on ranges provided by Hinkle et al. (2003). Parameters with moderate and high correlations with conductivity were considered chemical signatures of treated effluent exposure.

b) Above in diffuser area and main basin; above during ice-cover in northwest arm.

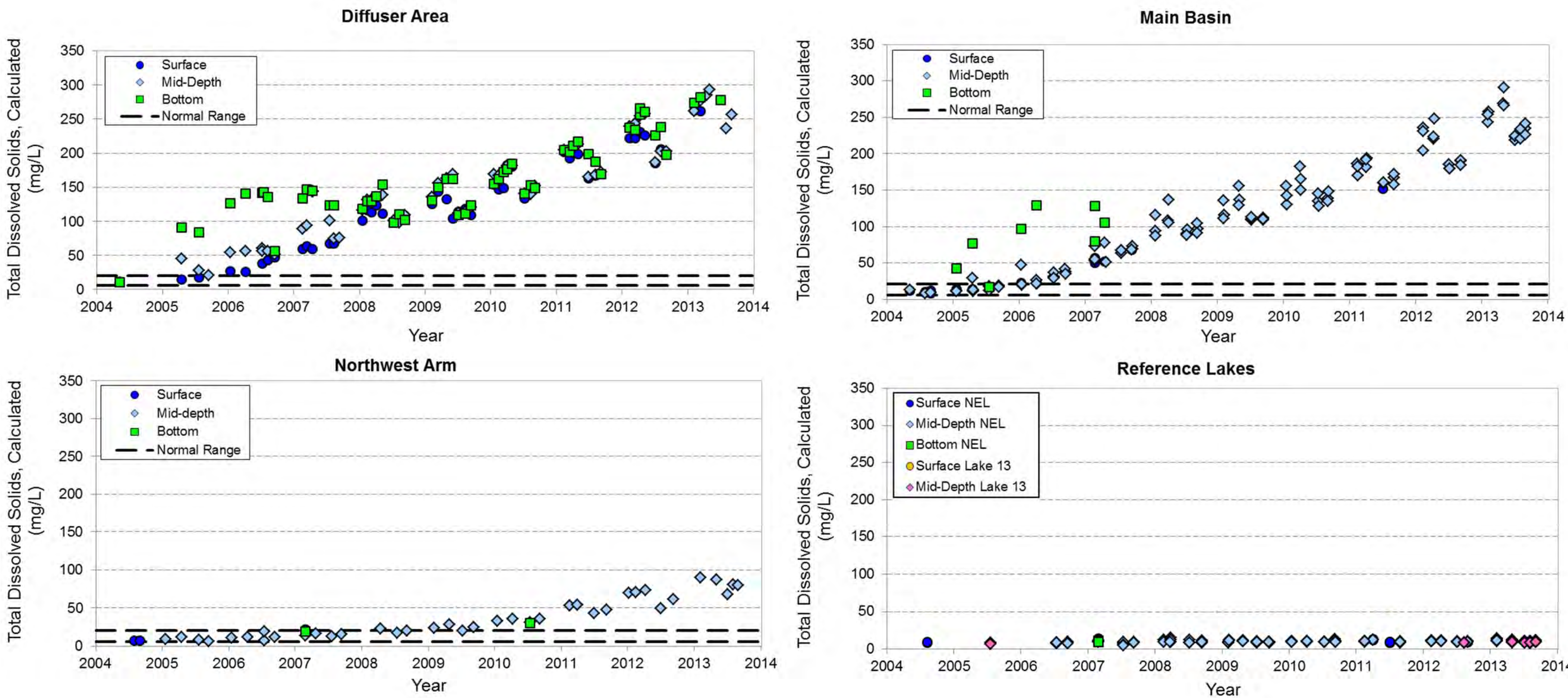
c) Above in diffuser area and main basin only.

d) The average concentration in Snap Lake was higher than the reference lakes, as described in Section 3.4.5.

e) Above in diffuser area; above during ice-cover in main basin; northwest arm is within the normal range.

$CaCO_3$ = calcium carbonate; HCO_3 = bicarbonate; N = nitrogen; P = phosphorus; SiO_2 = silicate; Flett = Flett Research Limited; *r* = Pearson's correlation co-efficient; ↑ = an increasing trend; - = indicates no increasing or decreasing trend; ± = plus or minus; <=less than; >= greater than; N/A = not available because normal ranges in field profile data were not calculated.

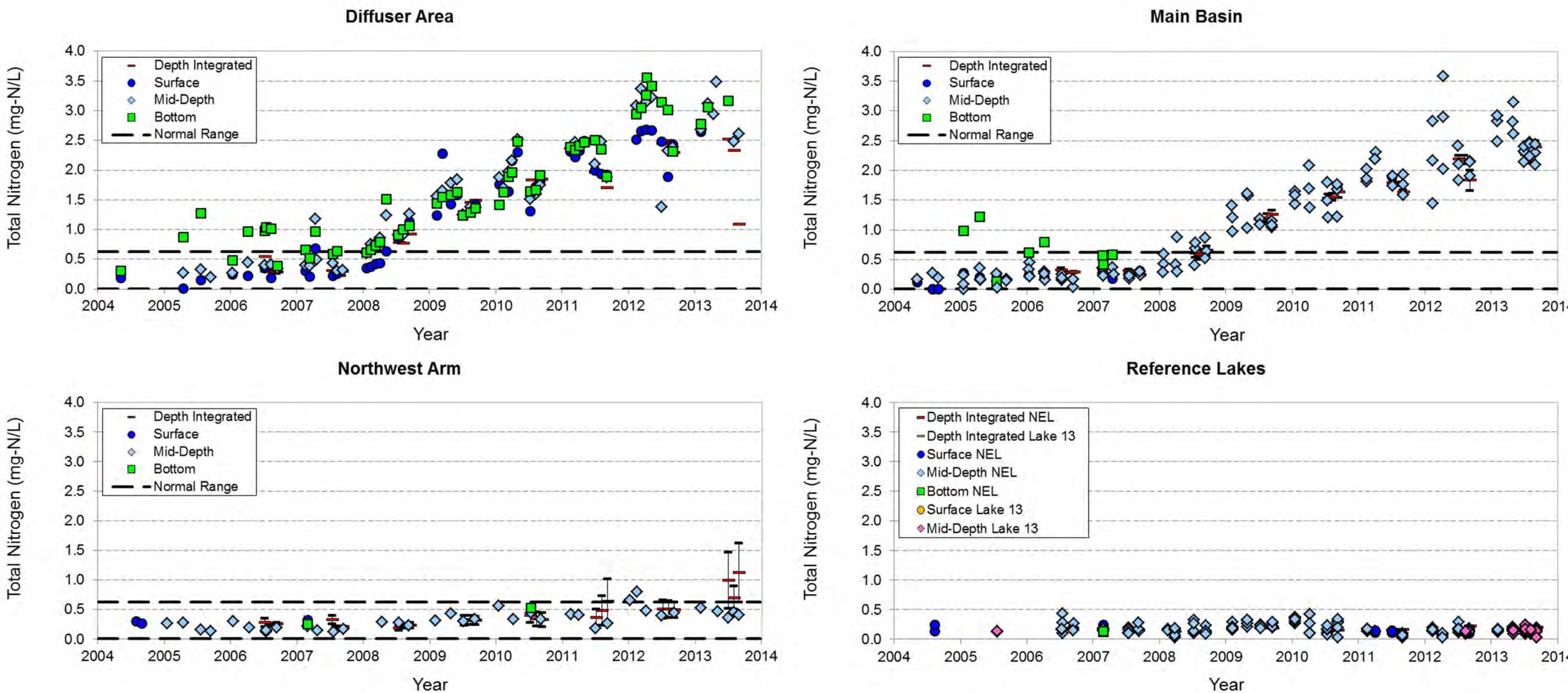
Figure 3-8 Total Dissolved Solids Concentrations in Snap Lake and the Reference Lakes, 2004 to 2013



Note: Normal range is based on data collected prior to 2004, with the upper and lower range calculated as the mean ± 2 standard deviations; data shown are from representative stations within Snap Lake including: Diffuser Area = SNAP 13 (2004 to April 2006) and SNP 02-20e (July 2006 to 2013); Main Basin = SNAP09,

SNAP05 and SNAP08 (2004 to 2013); Northwest Arm = SNAP02 (2004 to April 2006) and SNAP02A (July 2006 to 2013); Reference Lakes = NEL01 to NEL05 and LK13-01 to LK13-05; SNP = Surveillance Network Program; NEL = Northeast Lake, LK13 = Lake 13; and mg/L = milligrams per litre.

Figure 3-9 Total Nitrogen Concentrations in Snap Lake and the Reference Lakes, 2004 to 2013

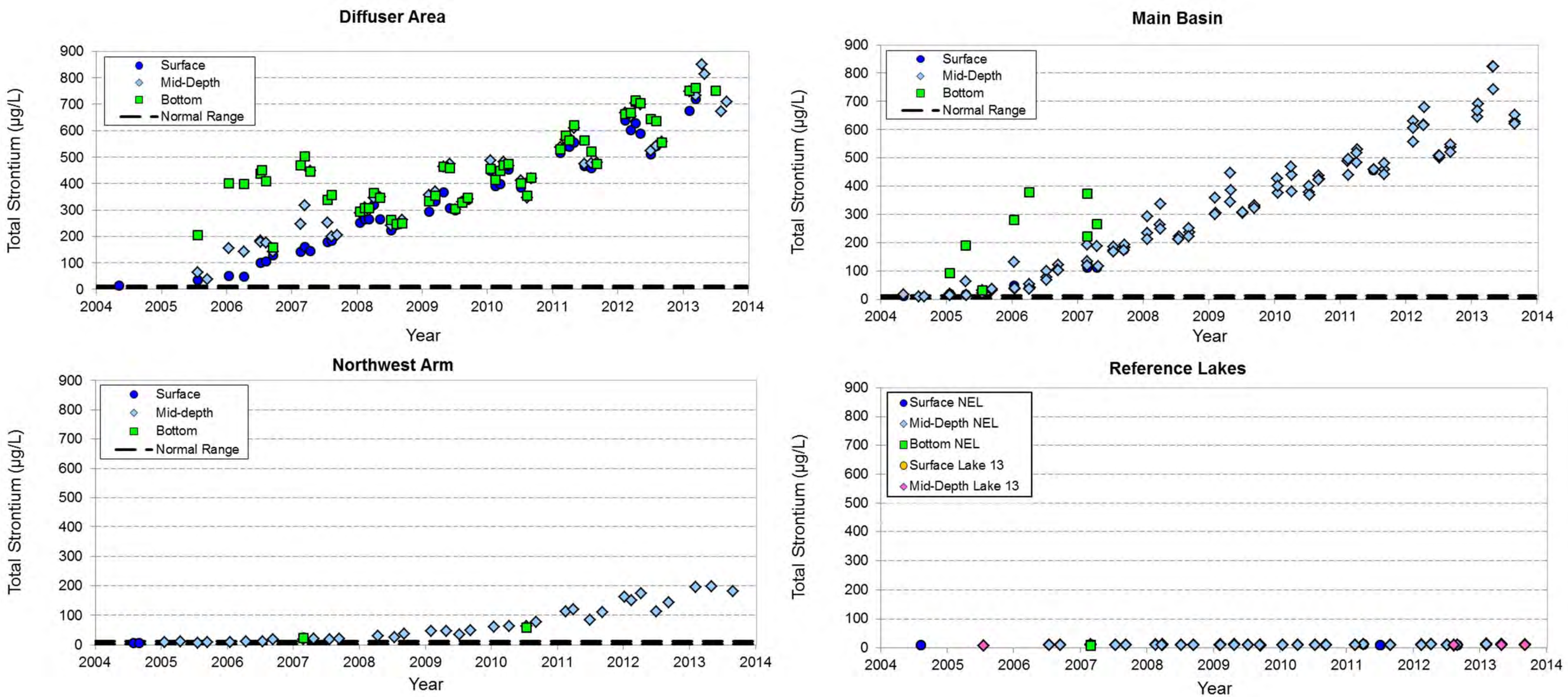


Note: Normal range is based on data collected prior to 2004, with the upper and lower range calculated as the mean ± 2 standard deviations; data shown are from representative stations within Snap Lake including: Diffuser Area = SNAP 13 (2004 to April 2006) and SNP 02-20e (July 2006 to 2013); Main Basin = SNAP09,

SNAP05 and SNAP08 (2004 to 2013); Northwest Arm = SNAP02 (2004 to April 2006) and SNAP02A (July 2006 to 2013); Reference Lakes = NEL01 to NEL05 and LK13-01 to LK13-05; Depth Integrated = monthly average of total nitrogen concentrations in the euphotic zone within each area of Snap Lake and the Reference Lakes (2006 to 2013), the black bars represent standard error around the average.

SNP = Surveillance Network Program; NEL = Northeast Lake, LK13 = Lake 13; mg-N/L = milligrams as nitrogen per litre.

Figure 3-10 Total Strontium Concentrations in Snap Lake and the Reference Lakes, 2004 to 2013

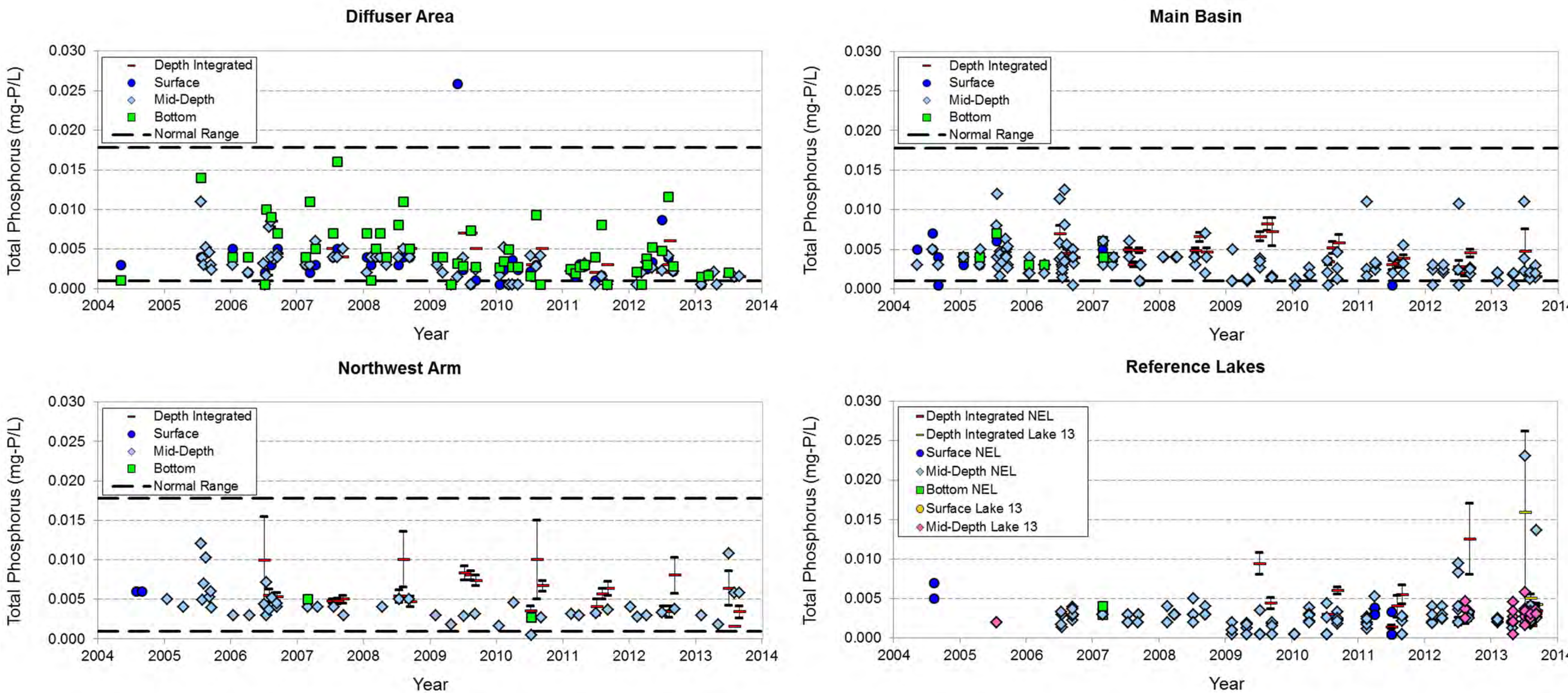


Note: Normal range is based on data collected prior to 2004, with the upper and lower range calculated as the mean ± 2 standard deviations; data shown are from representative stations within Snap Lake including: Diffuser Area = SNAP 13 (2004 to April 2006) and SNP 02-20e (July 2006 to 2013); Main Basin = SNAP09,

SNAP05 and SNAP08 (2004 to 2013); Northwest Arm = SNAP02 (2004 to April 2006) and SNAP02A (July 2006 to 2013); Reference Lakes = NEL01 to NEL05 and LK13-01 to LK13-05.

SNP = Surveillance Network Program; NEL = Northeast Lake, LK13 = Lake 13; µg/L = micrograms per litre.

Figure 3-11 Total Phosphorus Concentrations in Snap Lake and the Reference Lakes, 2004 to 2013

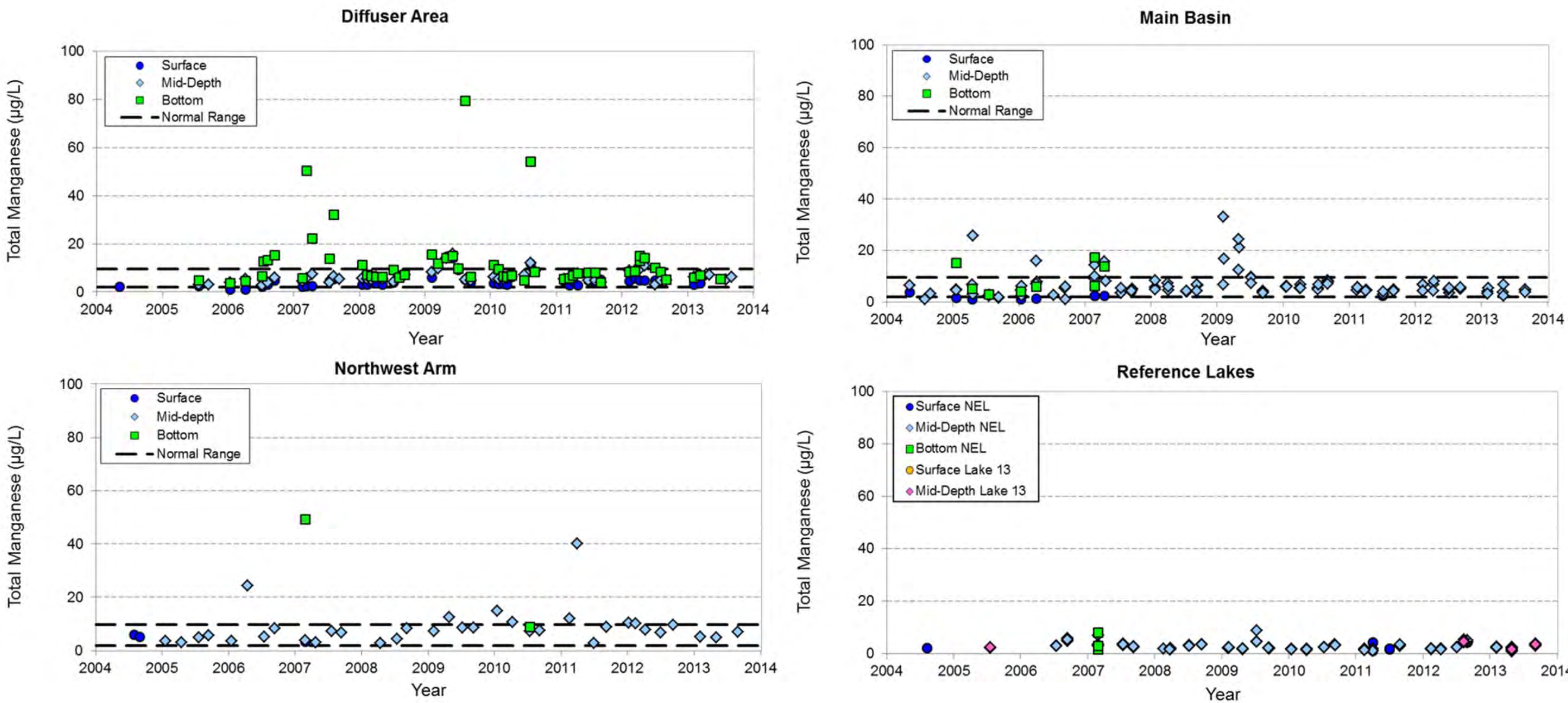


Note: Normal range is based on data collected prior to 2004, with the upper and lower range calculated as the mean ± 2 standard deviations; data shown are from representative stations within Snap Lake including: Diffuser Area = SNAP 13 (2004 to April 2006) and SNP 02-20e (July 2006 to 2013); Main Basin = SNAP09,

SNAP05 and SNAP08 (2004 to 2013); Northwest Arm = SNAP02 (2004 to April 2006) and SNAP02A (July 2006 to 2013); Reference Lakes = NEL01 to NEL05 and LK13-01 to LK13-05; Depth Integrated = monthly average of total phosphorus concentrations in the euphotic zone within each area of Snap Lake and the Reference Lakes (2006 to 2013), the black bars represent standard error around the average.

SNP = Surveillance Network Program; NEL = Northeast Lake, LK13 = Lake 13; mg-P/L = milligrams as phosphorus per litre.

Figure 3-12 Total Manganese Concentrations in Snap Lake and the Reference Lakes, 2004 to 2013



Note: Normal range is based on data collected prior to 2004, with the upper and lower range calculated as the mean ± 2 standard deviations; data shown are from representative stations within Snap Lake including: Diffuser Area = SNAP 13 (2004 to April 2006) and SNP 02-20e (July 2006 to 2013); Main Basin = SNAP09,

SNAP05 and SNAP08 (2004 to 2013); Northwest Arm = SNAP02 (2004 to April 2006) and SNAP02A (July 2006 to 2013); Reference Lakes = NEL01 to NEL05 and LK13-01 to LK13-05

SNP = Surveillance Network Program; NEL = Northeast Lake, LK13 = Lake 13; µg/L = micrograms per litre.

3.4.4.2 Comparison to EAR Predictions and 2013 Water Licence Renewal Application Model Predictions

The ranges in whole-lake average concentrations in Snap Lake were below the EAR[5] and the 2013 whole-lake average predictions for all parameters in 2013, with the exception of antimony, barium, and uranium. The 2013 whole-lake average for hexavalent chromium was less than the DL of 1 µg/L, which did not allow for a comparison to the EAR prediction of 0.8 µg/L. Barium and uranium, which were slightly above the 2013 whole-lake average prediction, increased in Snap Lake but either do not have an AEMP benchmark (i.e., barium) or are well below the benchmark (i.e., uranium). Uncertainties in model predictions are likely the cause of the higher than predicted concentrations; annual calibration updates to the model are expected to reduce these uncertainties over time. Antimony was also above the maximum life of Mine prediction from the 2013 modelling update; however, given the lack of a visually detectable increasing trend in Snap Lake (Appendix 3G, Figure 3G-27), the similarity between concentrations in Snap Lake and reference lakes (Appendix 3H, Figure 3H-34), and the known contamination issues with antimony (Appendix 3A, Section 3A1.2.2.4), the 2013 whole-lake average concentrations of antimony above predictions are more likely related to contamination issues rather than model uncertainties. To confirm the lack of temporal trend in antimony, statistical trend analysis was completed for this parameter (Seasonal Kendall Test in Section 3.4.4.2). Additional details regarding antimony, barium, uranium, and hexavalent chromium are provided in Section 3.4.4.2.

The comparisons of observed and predicted temporal trends focused on parameters with concentrations above AEMP benchmarks in 2013 (i.e., chloride, fluoride, nitrate) and also on TDS, which is a key indicator for Mine-related changes in Snap Lake water quality. The minimum field pH value and DO concentration were below the AEMP benchmark (Table 3-14). The EAR and the 2013 water quality model did not model pH levels in Snap Lake; therefore, temporal trends in pH could not be compared to predicted trends. Changes in DO profiles over time were compared to predicted changes in DO (Section 3.4.4.2).

[5] Environmental Assessment Report (EAR) predictions are based on maximum predicted whole-lake annual average concentrations (De Beers 2002). The range for Year 2013 was based on annual average concentrations using predicted daily whole-lake average concentrations for 2013 from the four modelled scenarios (i.e., Upper Bound Scenarios A and B and Lower Bound Scenarios A and B (De Beers [2013g]).

Table 3-14 Comparison of 2013 Snap Lake Water Quality to Environmental Assessment Report Predictions and 2013 Water Licence Renewal Application Model Predictions

Parameter	Units	AEMP Benchmarks and SSWQOs (Protection of Aquatic Life)[a]	Maximum Observed Concentration[b]	EAR Predictions[c]	2013 Predictions[d]		Observed Range in Whole Lake Average Concentration[e]
					Range for Year 2013	Maximum Life of Mine	
Field Parameters							
Dissolved Oxygen	mg/L	6.5, 9.5[f]	**5.7[g]**	-	-	-	0.3 to 18[h]
pH	unitless	6.5 to 9.0	**5.3[g]**	-	-	-	5.3 to 8.1[h]
Conventional Parameters							
Laboratory pH	unitless	6.5 to 9.0	6.8[g]	-	-	-	6.8 to 7.8[h]
Total dissolved solids, calculated (lab)	mg/L	-	301	350	224 to 363	1,685	228 to 284
Ions							
Chloride	mg/L	120 (218 to 388)[i]	134	137	99 to 163	770	104 to 128
Fluoride	mg/L	0.12 (2.46)[j]	0.23	-	0.16 to 0.21	0.46	0.13 to 0.18
Sodium	mg/L	-	33	-	24 to 40	193	24 to 31
Calcium	mg/L	-	61	88	46 to 75	352	46 to 57
Magnesium	mg/L	-	7.7	9	5.3 to 7.3	17	5.9 to 7.2
Sulphate	mg/L	-	26	-	19 to 29	115	20 to 24
Nutrients							
Nitrate, as N	mg-N/L	2.93 (4.1 to 16.4)[k]	3.04	5.87 / 6	2.06 to 2.83	8.39	2.05 to 2.67
Nitrite, as N	mg-N/L	0.06	0.027	-	-	-	-
Ammonia, as N	mg-N/L	0.51 to 125 [l]	0.3	1.23 / 1.1	0.20 to 0.41	1.5	0.07 to 0.26
Total phosphorus	mg-P/L	0.011[m]	-	0.013	-	-	<0.001 to 0.005
Total Metals							
Aluminum	µg/L	5 to 100[n]	8	-	4.3 to 6.9	15	2.2 to 6.6
Antimony	µg/L	-	3.77	-	0.20 to 0.30	0.56	0.07 to 0.8
Arsenic	µg/L	5	0.1	-	0.17 to 0.22	0.29	0.09 to 0.11
Barium	µg/L	-	29	-	18 to 23	34	19 to 26
Boron	µg/L	1500	69	-	-	-	-
Cadmium	µg/L	0.36	0.016	0.058	-	-	0.003 to 0.003
Chromium	µg/L	8.9	0.18	-	-	-	-
Hexavalent chromium	µg/L	2.1	1.3	0.8	-		<1.0
Copper	µg/L	7.9 (2.1 to 8.1[o])	1.1	2.2	0.9 to 1.2	2.14	0.4 to 0.46
Iron	µg/L	300	12	-	-	-	-
Lead	µg/L	1 to 6.96[p]	0.03	0.58	-	-	0.01 to 0.01
Lithium	µg/L	-	14	-	11 to 22	152	8 to 11
Manganese	µg/L	-	14	19	-	-	4.3 to 5.7
Mercury (Flett)	µg/L	0.026	0.002	-	-	-	-
Molybdenum	µg/L	73	1.73	-	-	-	-
Nickel	µg/L	45.3 to 153[p]	2.64	8.1	-	-	1.1 to 2.3
Selenium	µg/L	1	0.1	0.42	-	-	0.02 to 0.03
Silver	µg/L	0.1	0.006	-	-	-	-

Table 3-14 Comparison of 2013 Snap Lake Water Quality to Environmental Assessment Report Predictions and 2013 Water Licence Renewal Application Model Predictions

Parameter	Units	AEMP Benchmarks and SSWQOs (Protection of Aquatic Life)[(a)]	Maximum Observed Concentration[(b)]	EAR Predictions[(c)]	2013 Predictions[(d)]		Observed Range in Whole Lake Average Concentration[(e)]
					Range for Year 2013	Maximum Life of Mine	
Total Metals (Continued)							
Strontium	µg/L	-	850	-	607 to 938	3,890	665 to 808
Thallium	µg/L	0.8	<0.01	-	-	-	-
Uranium	µg/L	15	0.29	-	0.16 to 0.22	0.65	0.18 to <u>0.27</u>
Zinc	µg/L	30	3	-	2.2 to 2.8	4.27	0.4 to 1.7

a) AEMP benchmarks are: Water Quality Guidelines (WQGs) from the Canadian Council of Ministers of the Environment (CCME) (1999) and site-specific Environment Assessment Report (EAR) benchmarks from De Beers (2002). Recommended SSWQOs are provided in parentheses. Additional detail on benchmarks is provided in Table 3-10.

b) Observed concentrations within the 2013 reporting period (November 1, 2013 to October 31, 2013). The range in whole-lake average concentrations of total phosphorus is also presented. Bold values are above relevant benchmarks.

c) Environmental Assessment Report (EAR) predictions are based on maximum predicted whole-lake annual average concentrations (De Beers 2002). For EAR predictions of nitrate and ammonia, the initial value is a simulated summer average concentration provided in the supplemental information to the EAR (De Beers 2003). The latter value is a maximum predicted whole-lake average concentration as presented in the EAR (De Beers 2002).

d) The "Range for Year 2013" was based on annual average concentrations using predicted daily whole-lake average concentrations in Snap Lake for 2013 from the four modelled scenarios (i.e., Upper Bound Scenarios A and B and Lower Bound Scenarios A and B (De Beers [2013g]). The "Maximum Life of Mine" values were based on maximum predicted whole-lake average concentrations for the duration of operations (De Beers 2013g). The whole-lake average concentration calculations excluded the northwest arm.

e) Observed range in whole-lake average concentrations from 2013, which excludes northwest arm stations. Values above the maximum predicted whole-lake average annual concentrations from the EAR (De Beers 2002) or above the maximum predicted whole-lake average concentrations from the 2013 model predictions (De Beers 2013b) are <u>underlined</u>.

f) Lowest acceptable dissolved oxygen concentration for cold-water biota is 9.5 mg/L for early life stages, 6.5 mg/L for other life stages.

g) Minimum observed value within the 2013 reporting period (November 1, 2013 to October 31, 2013).

h) Observed range from the 2013 reporting period (February 1, 2013 to October 1, 2013), rather than whole-lake average.

i) The chloride SSWQO was developed as part of the TDS Response Plan (De Beers 2013h). The SSWQO range was based on the minimum hardness of 37 mg/L observed in Snap Lake during the 2013 reporting period and a maximum hardness of 160 mg/L. Although maximum hardness in Snap Lake was 185 mg/L in 2013, a hardness relationship with toxicity for chloride was not applicable for hardness values above 160 mg/L (Elphick et al. 2011).

j) The fluoride SSWQO was developed as part of the TDS Response Plan (De Beers 2013h, i).

k) The nitrate SSWQO was developed as part of Nitrate Response Plan (De Beers 2013j). The SSWQO range was based on the minimum hardness 37 mg/L observed in Snap Lake during the 2013 reporting period and a maximum hardness of 160 mg/L. Although maximum hardness in Snap Lake was 185 mg/L, a hardness relationship with toxicity for nitrate was not applicable for hardness values above 160 mg/L (Rescan 2012).

l) The ammonia WQG is pH and water temperature dependent. The CCME recommended that the guideline values falling into the range less than 5 degrees Celsius (°C) and greater than pH of 10 should be used with caution because the lack of toxicity data to accurately determine the toxicity effects at high and low extremes. Therefore, the range of the guideline shown is based on a range of the maximum field pH (8.1) and temperature (16.9°C) in Snap Lake over the 2013 reporting period and the lowest pH (6.0) and temperature (5°C) recommended in (CCME 1999). The guideline was calculated based on an individual pH and water temperature for each sample with the final value expressed as ammonia nitrogen.

m) The total phosphorus (TP) TP benchmark was derived from Wetzel (2001); mesotrophic conditions were defined by TP of 10.9 to 95.6 µg/L. The benchmark was based on the lower end of this range.

n) The aluminum WQG is pH dependent. The guideline shown is based on a range of field pH observed in Snap Lake during the 2013 reporting period (5.3 to 8.1). The WQG was calculated based on the individual pH for each sample.

o) The copper site-specific EAR benchmark was based on the minimum hardness of 37 mg/L and maximum of 185 mg/L which were observed in Snap Lake during the 2013 reporting period.

p) The lead and nickel WQGs are hardness dependent. The range of the WQGs shown was based on a range of hardness observed in Snap Lake during the 2013 reporting period (37 to 185 mg/L). The WQG was calculated based on the individual hardness for each sample.

N = nitrogen; Flett = Flett Research Limited; - = not applicable; < = less than; max = maximum; % = percent; µg/L = micrograms per litre; mg/L = milligrams per litre; mg-N/L=milligrams as nitrogen per litre; mg-P/L = milligrams as phosphorus per litre; °C = degrees Celsius; AEMP = Aquatic Effects Monitoring Program; and SSWQO = Site Specific Water Quality Objectives.

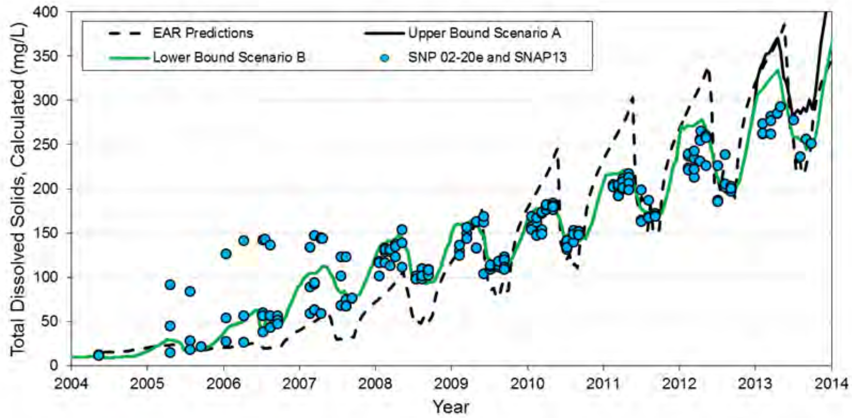
Total Dissolved Solids

Increasing trends in TDS concentrations were observed in all areas of Snap Lake (Figure 3-8). The largest increase in TDS was observed in the diffuser area and the main basin, where concentrations reached above 290 mg/L compared to the northwest arm where the maximum concentration at station SNAP02A was approximately 95 mg/L (Figure 3-8). Water quality in the northwest arm of Snap Lake has been the least influenced by treated effluent from the Mine because of the limited hydraulic connection of this area with the main basin of Snap Lake. However, an increasing trend in TDS concentrations has been evident in the northwest arm since 2008 (Figure 3-8), confirming an increasing treated effluent exposure in this area.

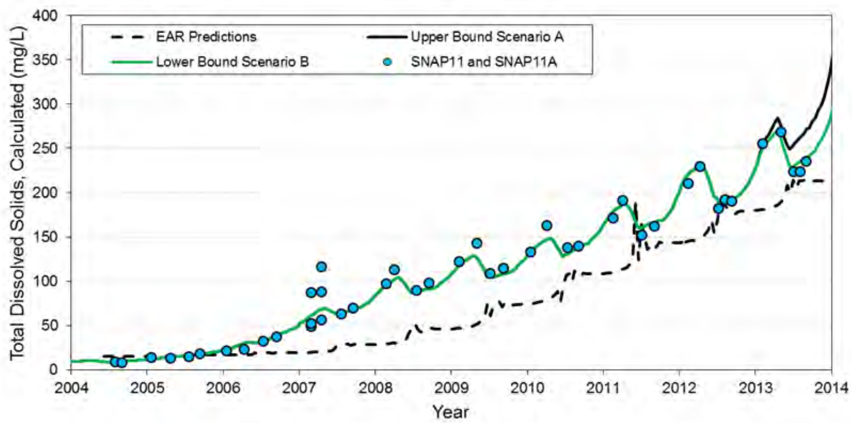
Concentrations of TDS in 2013 at the diffuser area, main basin, and Snap Lake outlet were overlain on the EAR and the 2013 model predictions in Figure 3-15. Concentrations of TDS within 200 m of the treated effluent discharge (i.e., the diffuser area) were below the EAR and the 2013 model predictions during ice-covered conditions, and between the EAR and the upper bound of the 2013 model predictions during open-water conditions (Figure 3-13, Panel a). Concentrations of TDS in the main basin at 2,000 m from the treated effluent discharge and at the Snap Lake outlet increased at a faster rate than EAR predictions but were consistent with the lower bound of the 2013 model predictions (Figure 3-13, Panels b and c). The 2013 whole-lake average TDS concentrations, which were below the Water Licence limit of 350 mg/L, were also consistent with the 2013 whole-lake average predictions (Figure 3-14), likely due to the similarly between cumulative measured and predicted TDS loadings (Figure 3-15).

Figure 3-13 Measured and Predicted Total Dissolved Solids Concentrations in Snap Lake

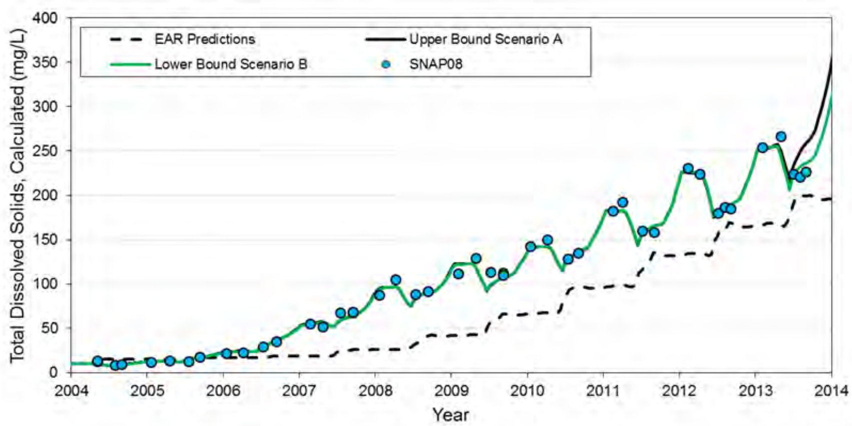
a. Diffuser Area (200 m from the Treated Effluent Discharge)



b. Main Basin (2,000 m from the Treated Effluent Discharge)



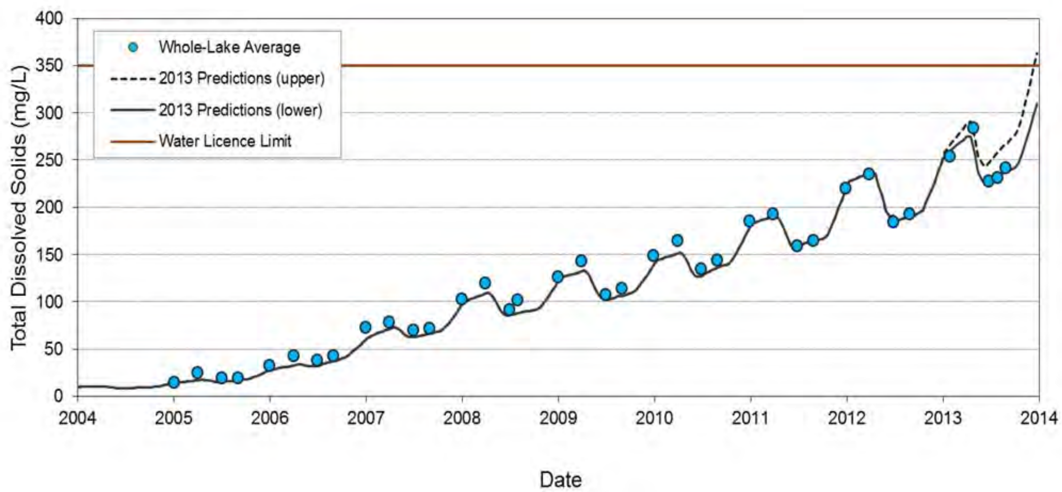
c. Snap Lake Outlet



Note: Data shown are from representative stations within Snap Lake: Diffuser Area = SNP 02-20e and SNAP13; Main Basin = SNAP11 and SNAP11A; Snap Lake Outlet = SNAP08; 2013 predictions represent the upper bound and lower bound scenarios (De Beers 2013b); EAR predictions are from De Beers (2002).

mg/L = milligrams per litre; m = metre.

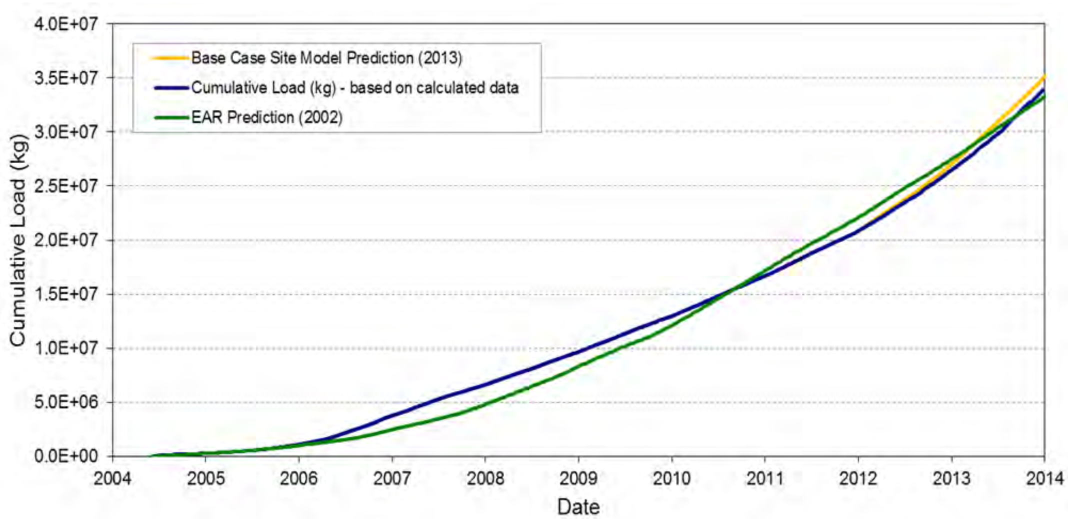
Figure 3-14 Measured and Predicted Whole-Lake Average Total Dissolved Solids Concentrations in Snap Lake



Note: 2013 whole-lake average predictions represent the upper and lower bound scenarios (De Beers 2013b); Water Licence limit issued in MV2011L2-0004 (MVLWB 2013a).

mg/L = milligrams per litre.

Figure 3-15 Measured and Predicted Cumulative Load of Total Dissolved Solids in Treated Effluent



kg = kilogram.

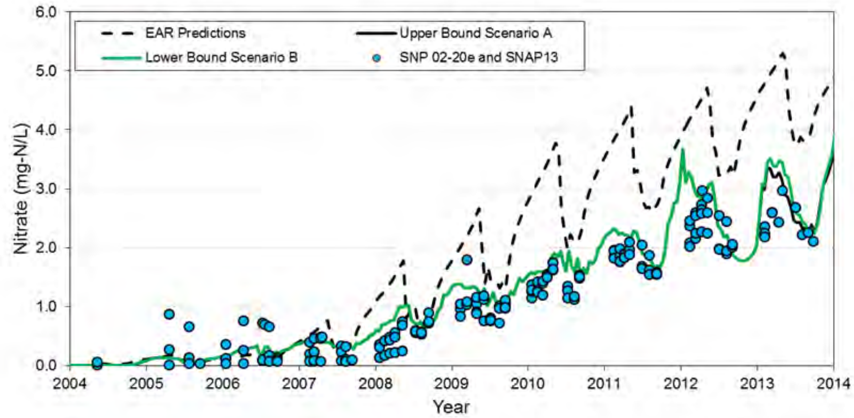
Nitrate

Nitrogen concentrations were expected to increase over time in the diffuser area and main basin because the treated effluent contains elevated concentrations of nitrogen. In particular, the treated effluent contains elevated concentrations of dissolved inorganic nitrogen, including nitrate and ammonia, from nitrogen-based explosives used in the mining process, and to a lesser degree, from the treated domestic waste water. Nitrate concentrations have increased since 2004 in the diffuser area and the main basin, and are beginning to increase in the northwest arm (Appendix 3G; Figure 3G-19).

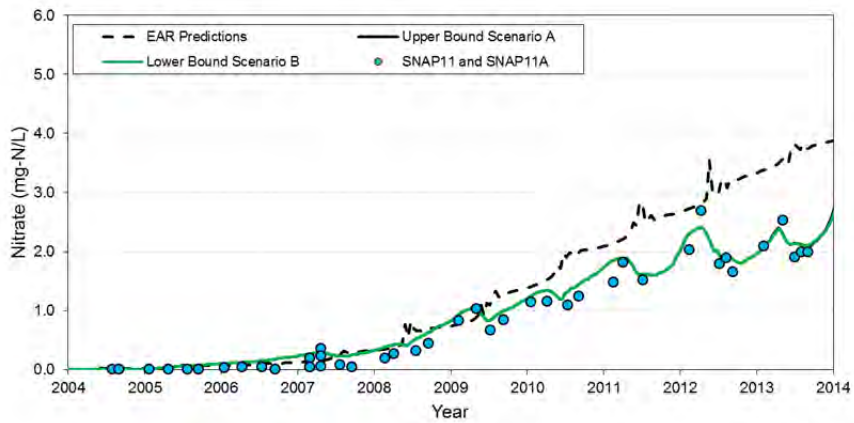
Measured and predicted nitrate concentrations in diffuser area, main basin, and at the Snap Lake outlet are presented in Figure 3-16. Since 2008, measured nitrate concentrations have been increasing in Snap Lake; however, the increase in nitrate concentrations has been slower than predicted in the EAR. As a result, measured nitrate concentrations in Snap Lake have remained below EAR model predictions, but are consistent with the 2013 model predictions. The whole-lake average measured nitrate concentrations in 2013 were below the CCME guideline of 2.93 mg-N/L (Figure 3-17).

Figure 3-16 Measured and Predicted Nitrate (as Nitrogen) Concentrations in Snap Lake

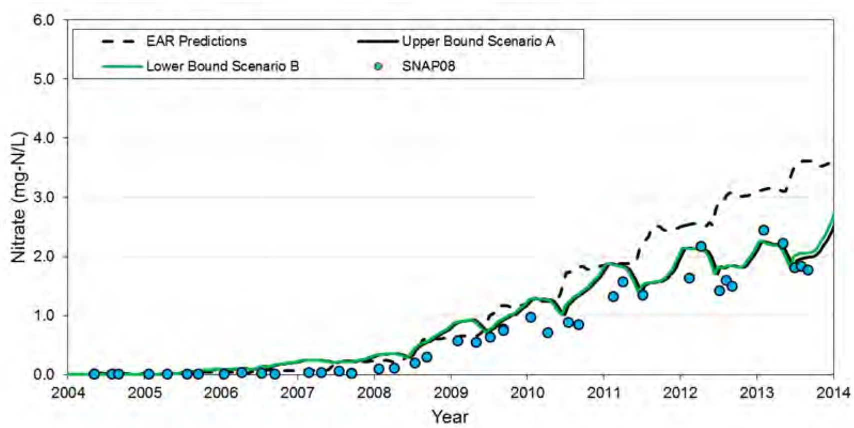
a. Diffuser Area (200 m from the Treated Effluent Discharge)



b. Main Basin (2,000 m from the Treated Effluent Discharge)



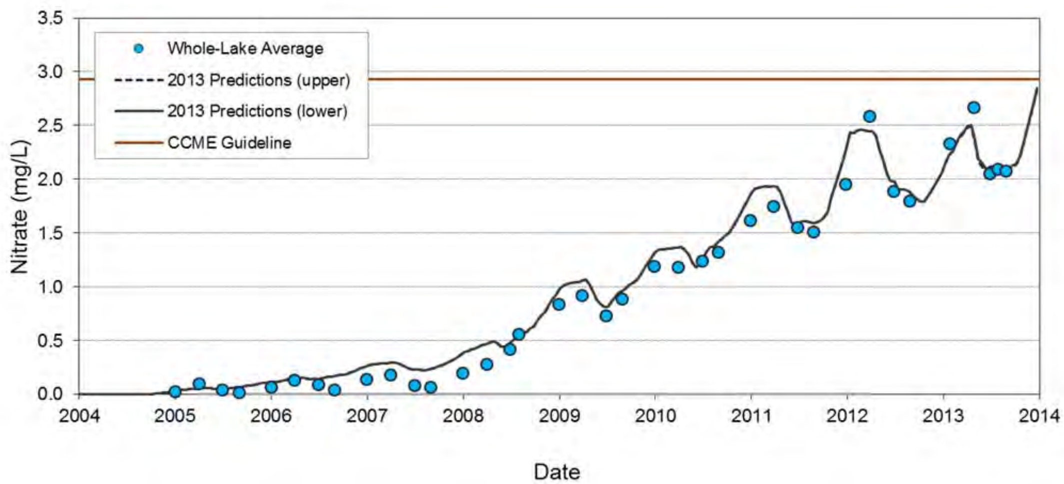
c. Snap Lake Outlet



Note: Data shown are from representative stations within Snap Lake: Diffuser Area = SNP 02-20e and SNAP13; Main Basin = SNAP 11 and SNAP 11A; Snap Lake Outlet = SNAP08; 2013 predictions represent the upper and lower bound scenarios (De Beers 2013b); EAR predictions are from De Beers (2003).

mg-N/L = milligrams as nitrogen per litre.

Figure 3-17 Observed and Predicted Whole-Lake Average Nitrate (as Nitrogen) Concentrations in Snap Lake



Note: 2013 whole-lake average predictions represent the upper and lower bound scenarios (De Beers 2013b); CCME guideline for nitrate is from CCME (1999).

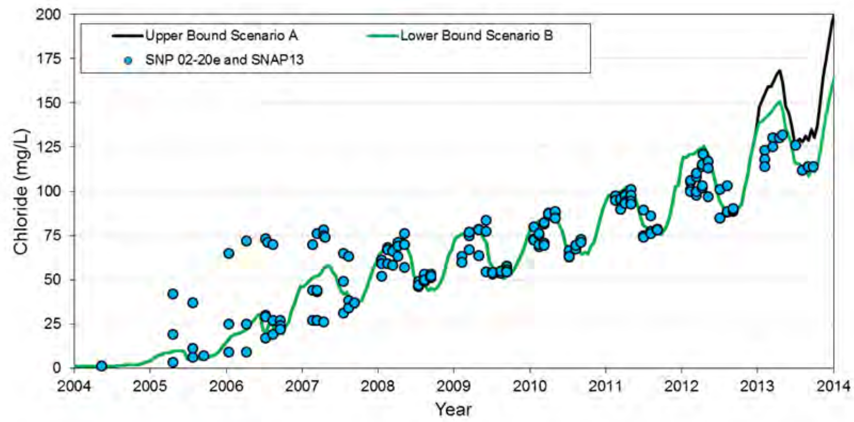
mg-N/L = milligrams as nitrogen per litre.

Chloride

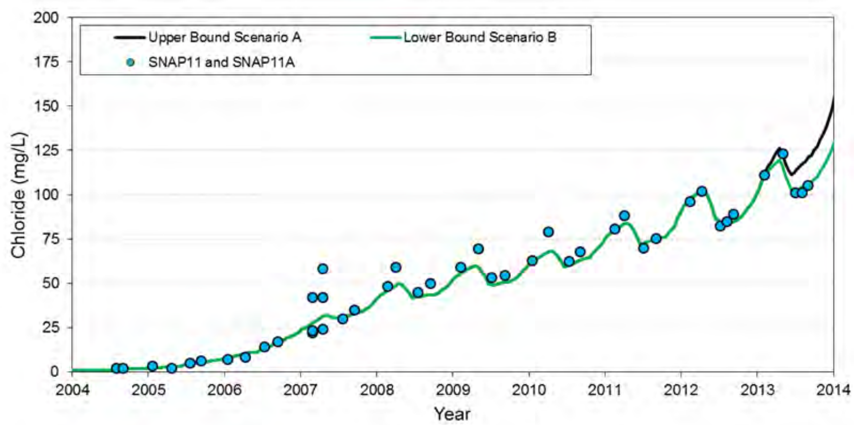
Measured and predicted chloride concentrations in the diffuser area, main basin, and at the Snap Lake outlet are presented in Figure 3-21. EAR model predictions are not available for chloride. Measured chloride concentrations have increased since 2005; concentrations in the diffuser area were slightly below the 2013 model predictions during ice-covered conditions and were consistent with the lower bound of the 2013 model predictions during open-water conditions (Figure 3-18, Panel a). Measured chloride concentrations at 2,000 m from the treated effluent discharge were within the 2013 model predictions throughout 2013 (Figure 3-18, Panel b); measured chloride concentrations at the Snap Lake outlet were slightly above the 2013 model predictions during ice-covered conditions, but within the 2013 model predictions during open-water conditions (Figure 3-18, Panel c). Whole-lake average chloride concentrations were within the 2013 whole-lake average predictions throughout 2013 (Figure 3-19). As predicted in 2013, the measured whole-lake average during ice-covered conditions in May 2013 was above the CCME guideline of 120 mg/L (Figure 3-19).

Figure 3-18 Measured and Predicted Chloride Concentrations in Snap Lake

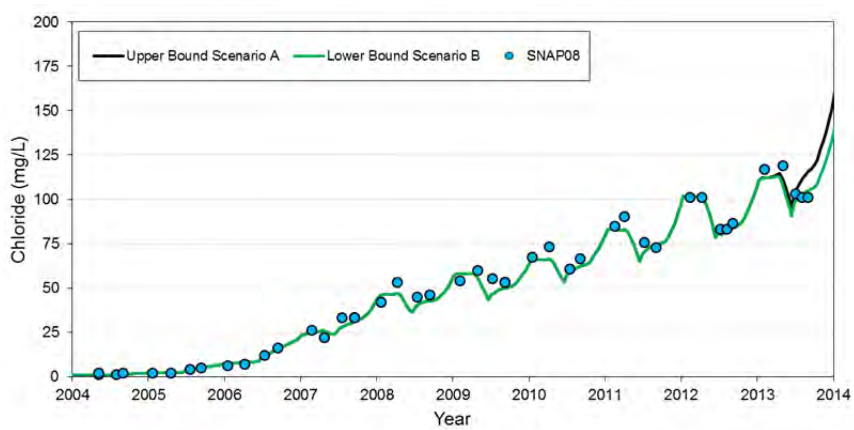
a. Diffuser Area (200 m from the Treated Effluent Discharge)



b. Main Basin (2,000 m from the Treated Effluent Discharge)



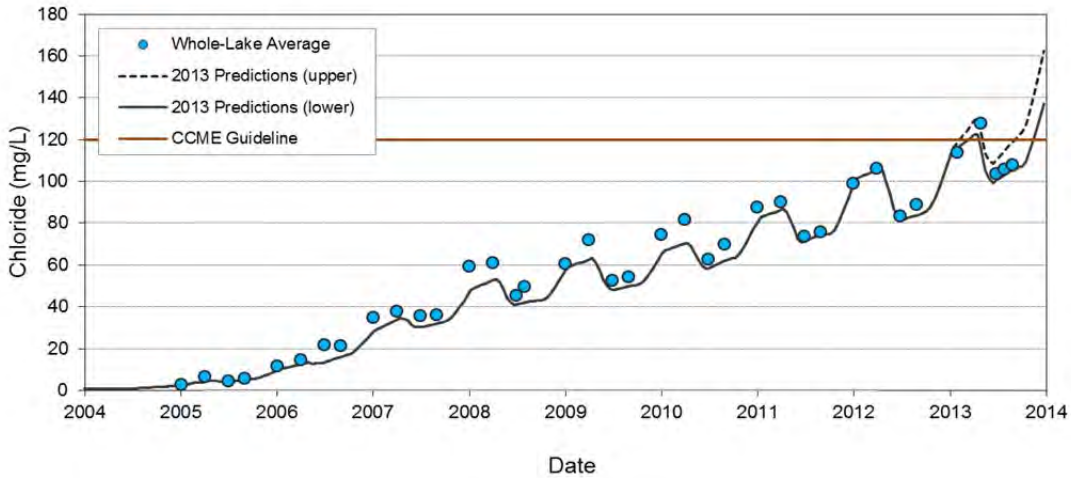
c. Snap Lake Outlet



Note: Data shown are from representative stations in Snap Lake: Diffuser Area = SNP 02-20e and SNAP13; Main Basin = SNAP11 and SNAP11A; Snap Lake Outlet = SNAP08; 2013 predictions represent the upper bound and lower bound scenarios (De Beers 2013b).

mg/L = milligrams per litre; m = metre.

Figure 3-19 Observed and Predicted Whole-Lake Average Chloride Concentrations in Snap Lake



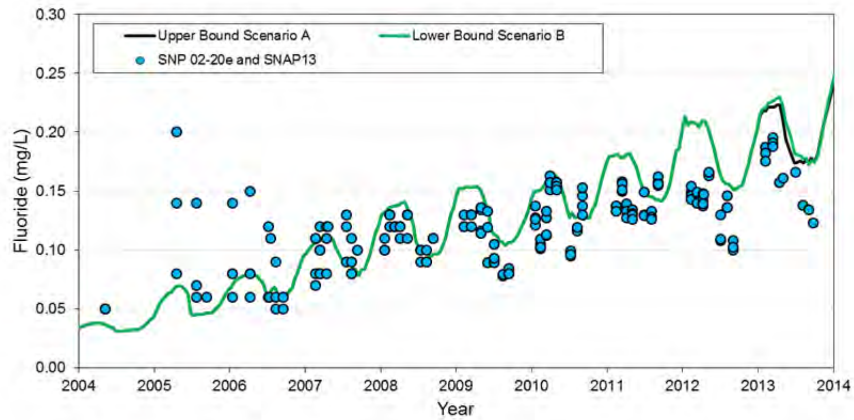
Note: 2013 predictions represent the upper and lower bound scenarios (De Beers 2013b); CCME guideline for chloride is from CCME (1999).
mg/L = milligrams per litre.

Fluoride

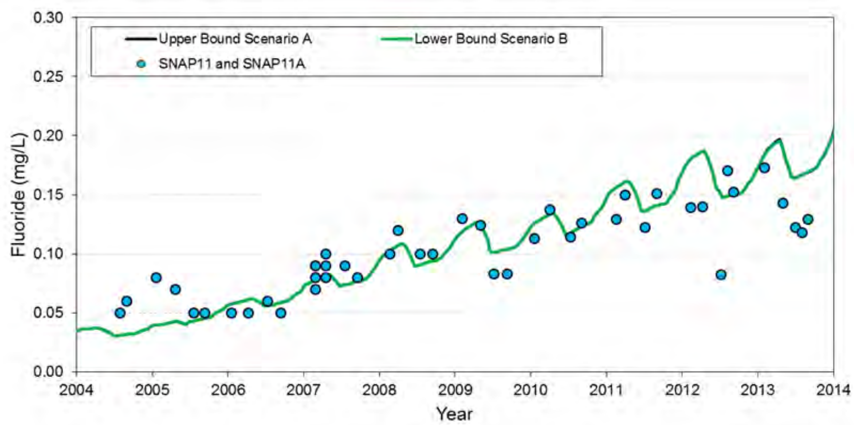
Measured and predicted fluoride concentrations in diffuser area, main basin, and at the Snap Lake outlet are presented in Figure 3-20. Measured fluoride concentrations have been increasing in the diffuser area and main basin since 2005 and continued to increase in 2013. Measured fluoride concentrations in the diffuser area, at 2,000 m from the treated effluent discharge, and at the Snap Lake outlet were below the 2013 model predictions throughout 2013 (Figure 3-20); EAR model predictions are not available for fluoride. Whole-lake average fluoride concentrations were above the CCME guideline of 0.12 mg/L (Figure 3-21) but remained below the recommended SSWQO for fluoride.

Figure 3-20 Measured and Predicted Fluoride Concentrations in Snap Lake

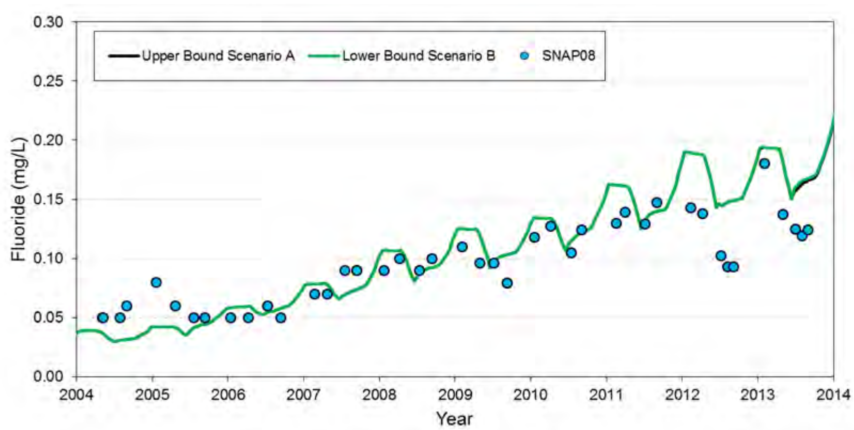
a. Diffuser Area (200 m from the Treated Effluent Discharge)



b. Main Basin (2,000 m from the Treated Effluent Discharge)



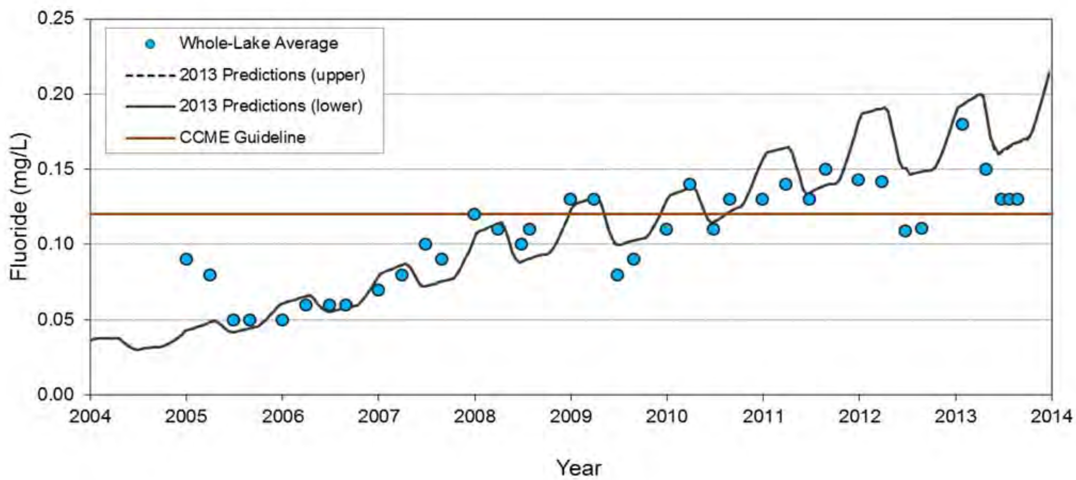
c. Snap Lake Outlet



Note: Data shown are from representative stations in Snap Lake: Diffuser Area = SNP 02-20e and SNAP13; Main Basin = SNAP11 and SNAP11A; Snap Lake Outlet = SNAP08; 2013 predictions represent the upper bound and lower bound scenarios (De Beers 2013b); EAR predictions are from De Beers (2002).

mg/L = milligrams per litre; m = metre.

Figure 3-21 Observed and Predicted Whole-Lake Average Fluoride Concentrations in Snap Lake



Note: 2013 predictions represent the upper and lower bound scenarios (De Beers 2013b); CCME guideline for fluoride is from CCME (1999).
mg/L = milligrams per litre.

Parameters above Predictions

The maximum whole-lake average for total barium and uranium concentrations were 26 µg/L and 0.27 µg/L, respectively, and were above the 2013 whole-lake average predictions (Table 3-14). Barium has increased in the diffuser area and throughout the main basin since 2005, and in the northwest arm since 2009 (Appendix 3G). Uranium has increased in the diffuser area and throughout the main basin since 2009, but is not increasing in the northwest arm (Appendix 3G). The divergence in observed and predicted barium and uranium concentrations in Snap Lake is likely due to model uncertainties; the divergence will continue to be reviewed and will be investigated if necessary (De Beers 2013b).

The maximum observed whole-lake average total antimony concentration of 0.8 µg/L was above both the 2013 and life of Mine whole-lake average predictions (Table 3-14). Increases in antimony have not been observed in Snap Lake (Appendix 3G, Figure 3G-27); Snap Lake and reference lake concentrations are similar (Appendix 3H, Figure 3H-34). Antimony contamination was identified in the blank samples, and antimony concentrations in the 2013 AEMP were qualified as having a high degree of uncertainty (Appendix 3A). Follow-up QA/QC investigations for antimony recommended for 2014 may result in further qualification of antimony data. Therefore, the differences between observed and predicted antimony concentrations are likely due to sample contamination issues. To confirm that antimony is not increasing in Snap Lake, a Seasonal Kendall test was completed (Section 3.4.4.2).

Two samples collected in 2013 were above the maximum whole-lake average EAR prediction of 0.8 µg/L for chromium (De Beers 2002). The range in whole-lake average concentration was less than 1.0 mg/L, which is a result of the limitation of the laboratory detection limit (DL). Samples collected in 2013 in Snap Lake were analyzed for hexavalent chromium using the best available DL of 1 µg/L

(Evaristo-Cordero 2013; Sahni 2013). The EAR predicted that elevated concentrations of hexavalent chromium would occur within 1% of the volume of Snap Lake. Because the values detected in the two samples were near the DL, they are considered an anomaly and not an indication of hexavalent chromium in groundwater seepage to the underground Mine as outlined in the EAR (De Beers 2002).

Maximum concentrations of total aluminum and zinc were above the predicted whole-lake averages for total aluminum and zinc. To confirm that these maximum concentrations are not an indication of an unexpected increasing trend, Seasonal Kendall tests were completed for these parameters.

Seasonal Kendall Test

Six parameters were statistically tested for trends using the Seasonal Kendall test to confirm or reject the presence of increasing trends for the following reasons:

- to determine whether field pH values in Snap Lake are increasing, which could not be clearly identified in temporal plots, based on the observation that laboratory pH values are increasing in Snap Lake;
- to determine whether manganese is increasing in Snap Lake, which could not be clearly identified in temporal plots, based on the observation that 2013 Snap Lake concentrations are greater than 2013 reference lake concentrations;
- to assess whether the 2013 maximum observed concentrations of zinc and aluminum that were above whole-lake average predictions are not related to an increasing trend in these two parameters;
- to determine whether the 2013 whole-lake averages of antimony that were above whole-lake average predictions are not related to an increasing trend in antimony; and,
- to determine whether phosphorus concentrations in Snap Lake are increasing using phosphorus results collected from both the water quality and plankton components.

The results of the Seasonal Kendall test for select parameters are summarized in Table 3-15. Separate Seasonal Kendall tests were performed on the TP results from the water quality and plankton components.

The Seasonal Kendall test identified increasing trends in field pH at all depths (i.e., surface, mid-depth, and bottom) at the diffuser stations, and at mid-depth throughout the main basin of Snap Lake, and in the northwest arm (Table 3-15).

Based on the result of the Seasonal Kendall test, manganese has increased at locations closer to the diffuser, with the exception of bottom depths, but not at stations further from the diffuser. Increasing trends in manganese at the mid and surface depths at the diffuser stations, and at mid-depth at SNAP05 were statistically significant; however, a decreasing trend was identified at mid-depth at the Snap Lake outlet (Table 3-15). No trends were identified at the main basin station SNAP09 or in the northwest arm (Table 3-15). The inconsistent trends may be caused by the non-conservative behaviour of manganese in the water column; oxidation and reduction processes may have a greater influence on manganese concentrations in the water column relative to loadings from the treated effluent.

The Seasonal Kendall tests of aluminum, antimony, and zinc showed a lack of consistent temporal trends in all three parameters, indicating that these three metals are not increasing in Snap Lake. Decreasing trends in aluminum concentrations were identified at all depths (i.e., surface, mid-depth, and bottom) at the diffuser stations, and at mid-depth in the northwest arm (Table 3-15). Antimony concentrations were neither increasing nor decreasing in Snap Lake, with the exception of the Snap Lake outlet where a decreasing trend was identified (Table 3-15). Similar to antimony, concentrations of zinc were also neither increasing nor decreasing in Snap Lake, with the exception of station SNAP05 where there was an increasing trend (Table 3-15).

The results of the Seasonal Kendall test for phosphorus indicate that this nutrient is not increasing in Snap Lake. A decreasing trend in TP concentrations was identified in the samples collected as part of the water quality component at all depths (i.e., surface, mid-depth, and bottom) at the diffuser stations and at mid-depth in the main basin. No trends were identified in the northwest arm. The Seasonal Kendall test for TP concentrations collected as part of the plankton component did not identify an increasing nor decreasing trend, thus indicating that phosphorus concentrations have remained similar over time in the euphotic zone of Snap Lake (Table 3-15).

Possible explanations for the absence of an increasing trend in Snap Lake phosphorus concentrations are:

- Aquatic organisms may be rapidly taking up phosphorus released at the diffuser, and then dying off and settling to the bottom.
- The littoral zone may be intercepting the phosphorus before it can be measured in the water column during open-water conditions (Section 11.3).
- Phosphorus concentrations in Snap Lake are near laboratory DLs and, at these low concentrations, there is a greater degree of uncertainty in the laboratory reported concentrations, as described in the Nutrient Assessment (Appendix 3B). Small increases in phosphorus in the range of the level of uncertainty (approximately ± 0.002 mg-P/L) may not be detectable.

Table 3-15 Summary of Temporal Trends for Selected Parameters and Stations Using the Seasonal Kendall Test

Parameter	Lake Area (Representative Station)	Depth	n	Z Value at 95% Confidence[a]	P Value at 95% Confidence[a]	Significant Trend
Field measured pH (as a two-sided trend)	Diffuser (SNAP 13 and SNP 02-20e)	Surface	31	1.996	0.046	↑
		Mid	36	4.425	<0.001	↑
		Bottom	33	4.339	<0.001	↑
	Main Basin (SNAP05)	Mid	46	5.738	<0.001	↑
	Main Basin (SNAP08)	Mid	47	4.354	<0.001	↑
	Main Basin (SNAP09)	Mid	44	7.144	<0.001	↑
	Northwest Arm (SNAP02A)	Mid	22	2.532	0.011	↑
Total Aluminum (as a two-sided trend)	Diffuser (SNAP 13 and SNP 02-20e)	Surface	51	-3.376	0.001	↓
		Mid	63	-3.463	0.001	↓
		Bottom	51	-2.771	0.006	↓
	Main Basin (SNAP05)	Mid	46	0.205	0.838	-
	Main Basin (SNAP08)	Mid	50	-1.392	0.164	-
	Main Basin (SNAP09)	Mid	44	0.800	0.424	-
	Northwest Arm (SNAP02A)	Mid	33	-4.456	<0.001	↓
Total Antimony (as a two-sided trend)	Diffuser (SNAP 13 and SNP 02-20e)	Surface	51	1.016	0.309	-
		Mid	63	-1.067	0.286	-
		Bottom	48	1.049	0.294	-
	Main Basin (SNAP05)	Mid	39	0.200	0.842	-
	Main Basin (SNAP08)	Mid	50	-1.989	0.047	↓
	Main Basin (SNAP09)	Mid	39	-0.297	0.767	-
	Northwest Arm (SNAP02A)	Mid	30	1.505	0.132	-
Total Manganese (as a two-sided trend)	Diffuser (SNAP 13 and SNP 02-20e)	Surface	51	3.267	0.001	↑
		Mid	68	4.245	<0.001	↑
		Bottom	53	-0.105	0.917	-
	Main Basin (SNAP05)	Mid	46	3.672	<0.001	↑
	Main Basin (SNAP08)	Mid	50	-2.844	0.004	↓
	Main Basin (SNAP09)	Mid	44	0.725	0.468	-
	Northwest Arm (SNAP02A)	Mid	33	-0.211	0.833	-

Table 3-15 Summary of Temporal Trends for Selected Parameters and Stations Using the Seasonal Kendall Test

Parameter	Lake Area (Representative Station)	Depth	n	Z Value at 95% Confidence[a]	P Value at 95% Confidence[a]	Significant Trend
Total Zinc (as a two-sided trend)	Diffuser (SNAP 13 and SNP 02-20e)	Surface	51	-0.475	0.635	-
		Mid	68	1.617	0.106	-
		Bottom	53	-0.069	0.945	-
	Main Basin (SNAP05)	Mid	46	2.056	0.04	↑
	Main Basin (SNAP08)	Mid	50	0.812	0.417	-
	Main Basin (SNAP09)	Mid	44	0.638	0.523	-
	Northwest Arm (SNAP02A)	Mid	33	1.891	0.059	-
Total Phosphorus (as a two-sided trend)	Diffuser (SNAP 13 and SNP 02-20e)	Surface	51	-2.472	0.013	↓
		Mid	68	-3.300	0.001	↓
		Bottom	54	-3.056	0.002	↓
		Depth Integrated	6	-1.007	0.314	-
	Main Basin (SNAP05)	Mid	46	-3.085	0.002	↓
	Main Basin (SNAP05)	Depth Integrated	9	-0.419	0.675	-
	Main Basin (SNAP08)	Mid	50	-2.506	0.012	↓
	Main Basin (SNAP08)	Depth Integrated	27	-1.530	0.126	-
	Main Basin (SNAP09)	Mid	44	-3.424	0.001	↓
	Main Basin (SNAP09)	Depth Integrated	9	-0.535	0.593	-
	Northwest Arm (SNAP02A)	Mid	33	0.336	0.737	-
	Northwest Arm (SANP02 and SNAP02A)	Depth Integrated	16	0.408	0.684	-

Note: The Seasonal Kendall Test was run using SYSTAT 13.1 (SYSTAT 2009);

a) The critical Z-values associated with a two-sided 95% confidence interval are -1.96 and 1.96. The P-value associated with a 95% confidence interval is 0.05. If the Z-value is between -1.96 and 1.96 for a two-sided test, the P-value will be greater than 0.05 and the test concludes that no significant increasing or decreasing trend exists in the data.

↑ = an increasing trend; ↓ = a decreasing trend; - = no significant increasing or decreasing trend; % = percent; *P*= probability; and n = sample count.

Dissolved Oxygen

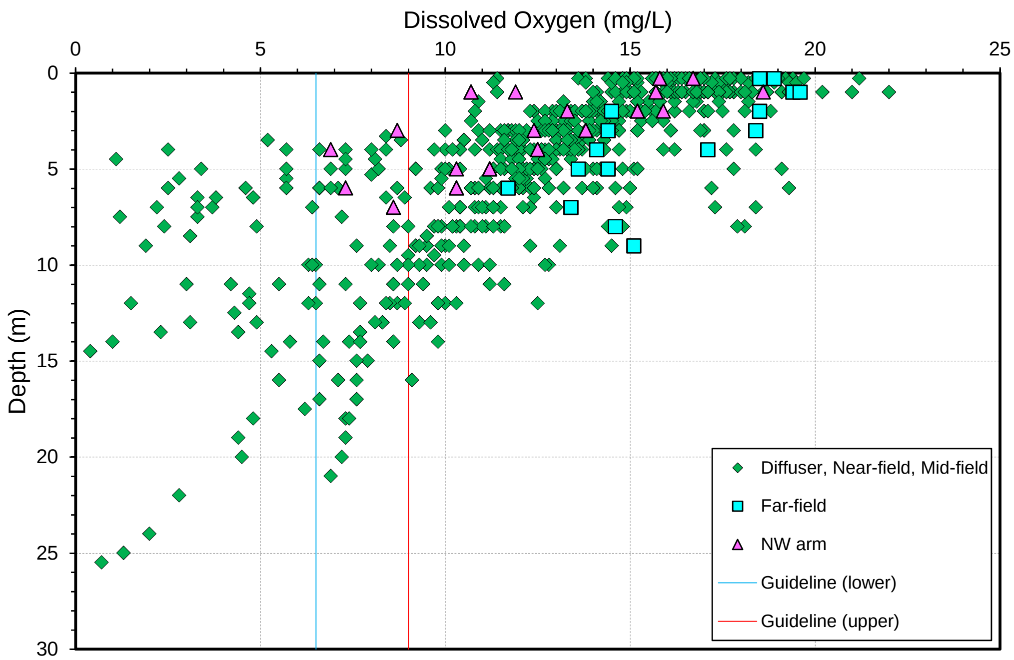
Vertical profiles of DO concentration during ice-cover between 1999 and 2013 are shown in Figure 3-22 (panels a to j). Profile data collected before treated effluent discharge to Snap Lake (1999 to 2004) were combined onto one graph (Figure 3-22, Panel a), and data collected during the period of treated effluent discharge are presented by year (Figure 3-22, panels b to j). A greater number of deeper stations were sampled in 2006 to 2013 compared to 2005 (Figure 3-22, panels b to j) because stations were relocated to deeper locations to allow for the assessment of DO conditions in deeper waters.

The concentration of DO in Snap Lake was predicted to decrease by 1.0 to 2.2 mg/L near the bottom of the lake during ice-covered conditions (De Beers 2002). The EAR also predicted that DO concentrations near the surface of the lake could decrease by up to 1 mg/L. Overall, near-bottom DO concentrations after 2004 have typically been greater than those prior to 2004 (Figure 3-22). Before the discharge began (1999 to 2004), DO concentrations decreased with depth to near 0 mg/L at deeper diffuser and main basin stations during ice-covered conditions (Figure 3-22, Panel a). In general, between 2005 and 2013, near-bottom DO concentrations at the diffuser and main basin stations during ice-covered conditions during discharge were higher than near-bottom DO concentrations at the same stations before discharge.

Anoxic conditions, when DO concentrations approached 0 mg/L, were measured near the lake bottom at the deepest diffuser station (SNP 02-20e) in 2007, but these conditions were not observed between 2008 and 2013 at this location. Low oxygen and anoxic conditions were observed near the bottom at some locations in the northwest arm between 2005 and 2013 (Figure 3-22, panels b to j).

Figure 3-22 Dissolved Oxygen Concentrations during Ice-Covered Conditions, 1999 to 2012

a. 1999 to 2004 – Pre-discharge



b. 2005

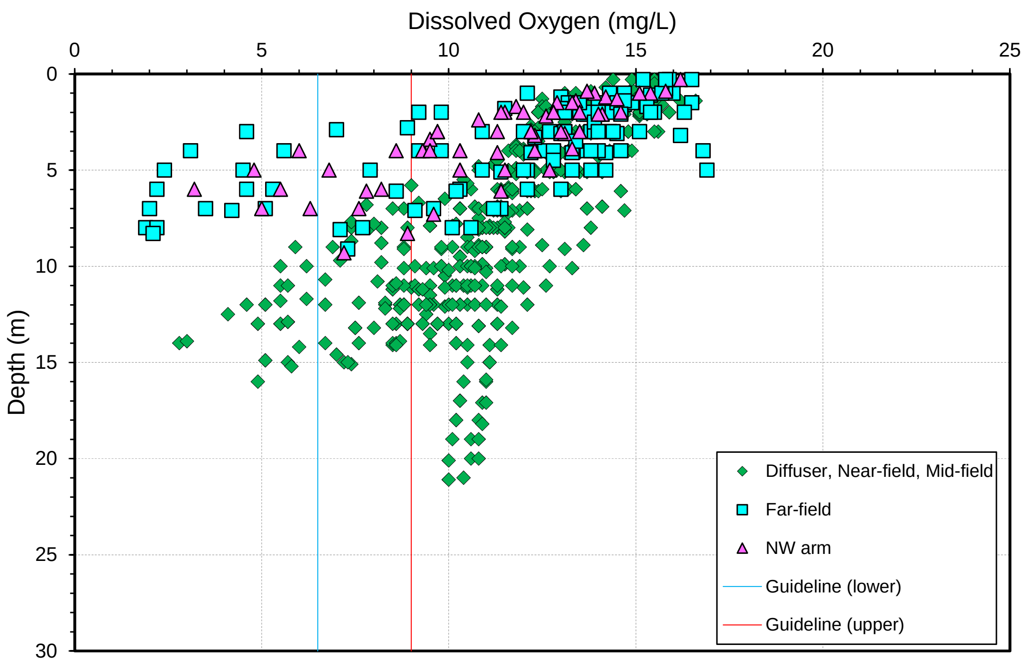
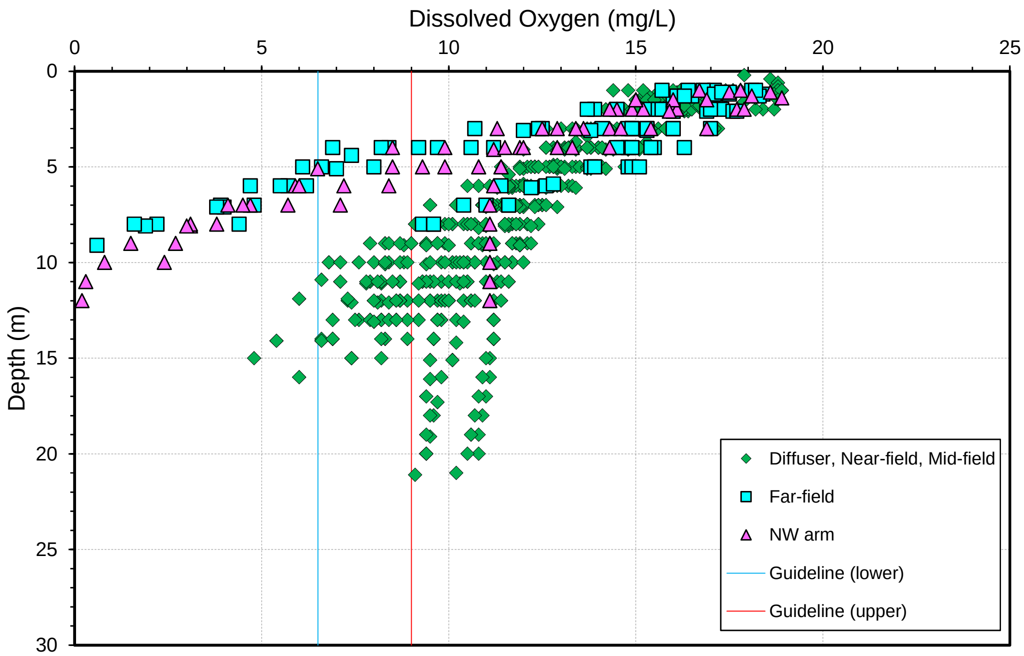


Figure 3 22 Dissolved Oxygen Concentrations during Ice-Covered Conditions, 1999 to 2012

c. 2006



d. 2007

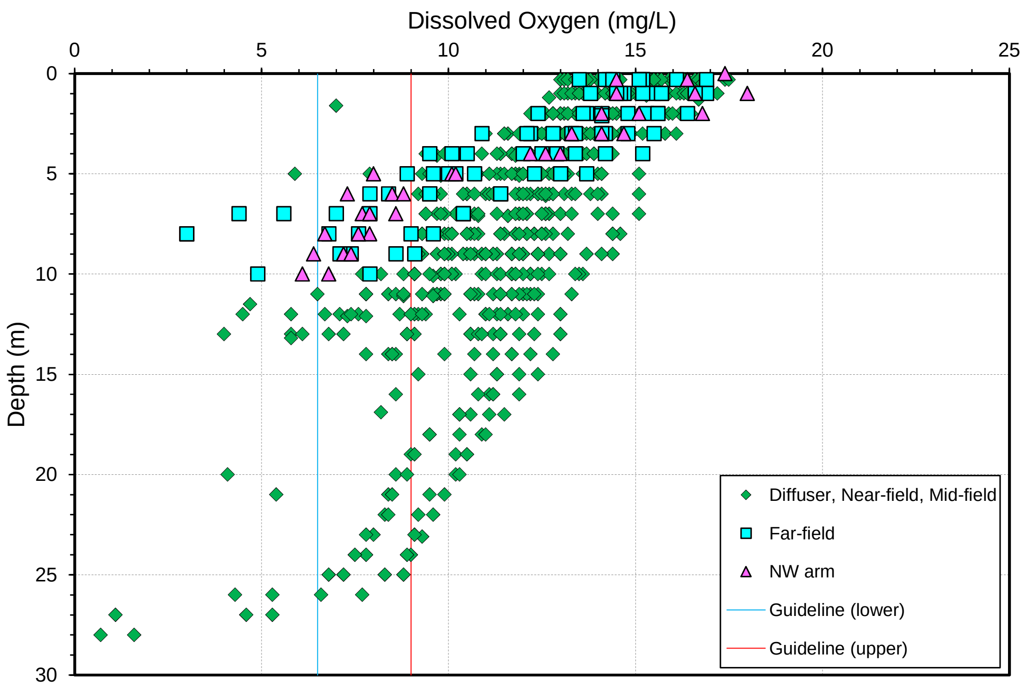
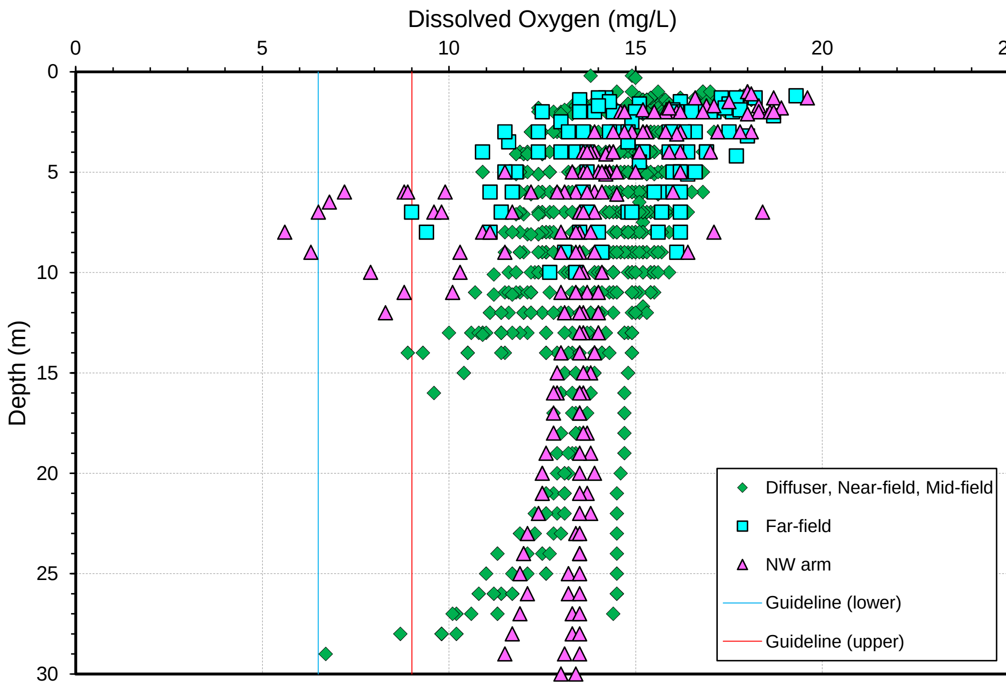


Figure 3 22 Dissolved Oxygen Concentrations during Ice-Covered Conditions, 1999 to 2012

e. 2008



f. 2009

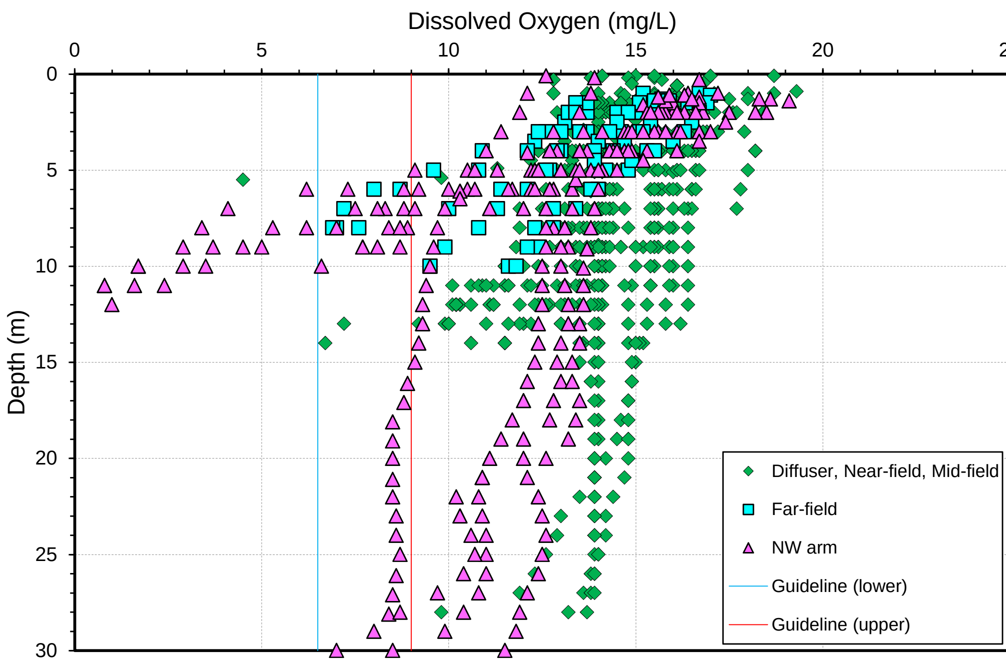
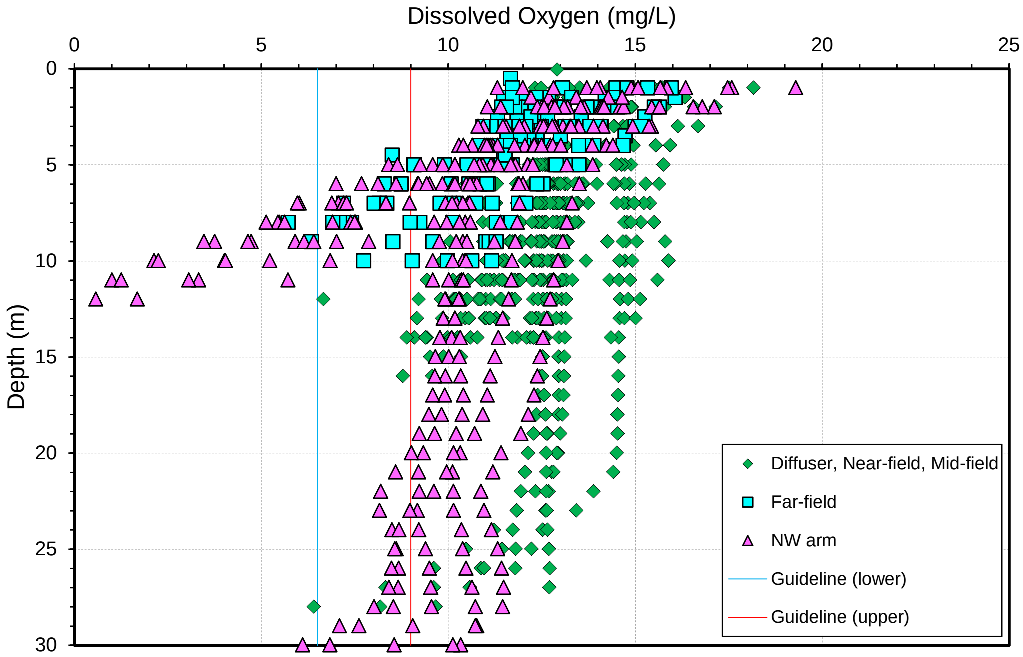


Figure 3 22 Dissolved Oxygen Concentrations during Ice-Covered Conditions, 1999 to 2012

g. 2010



h. 2011

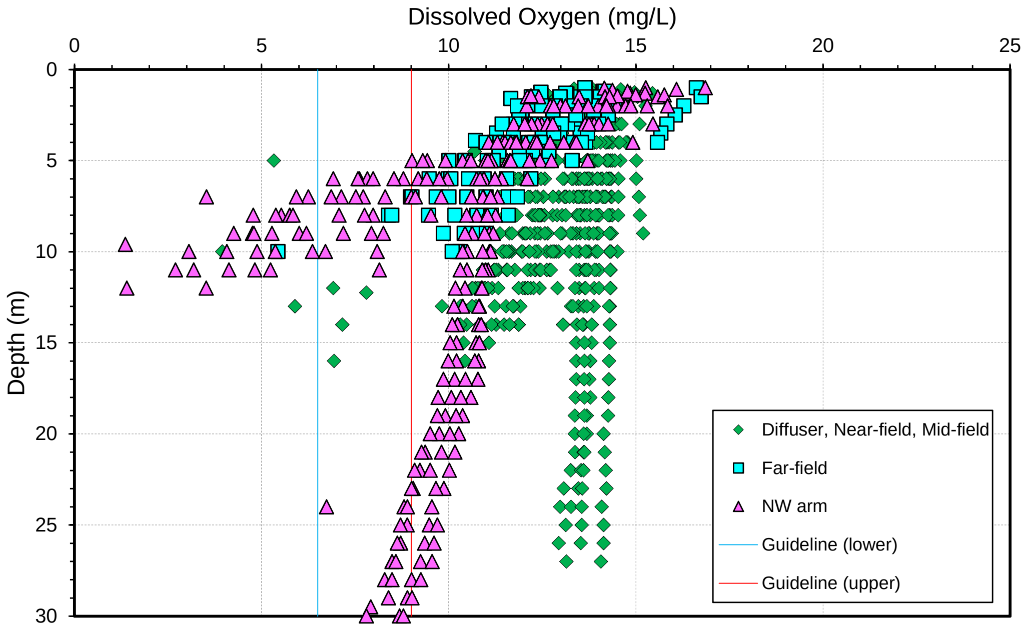
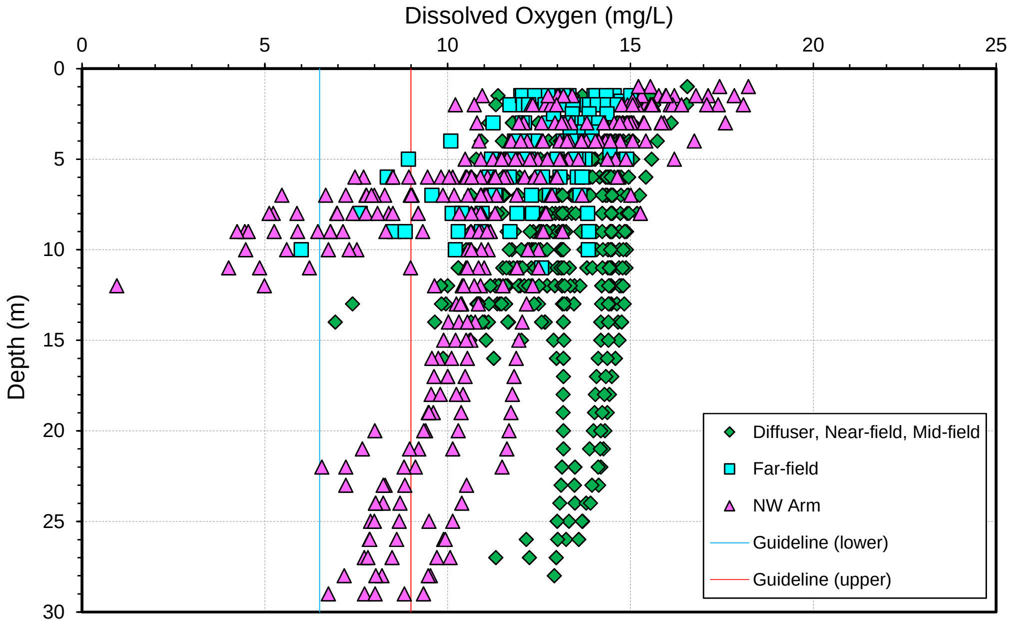
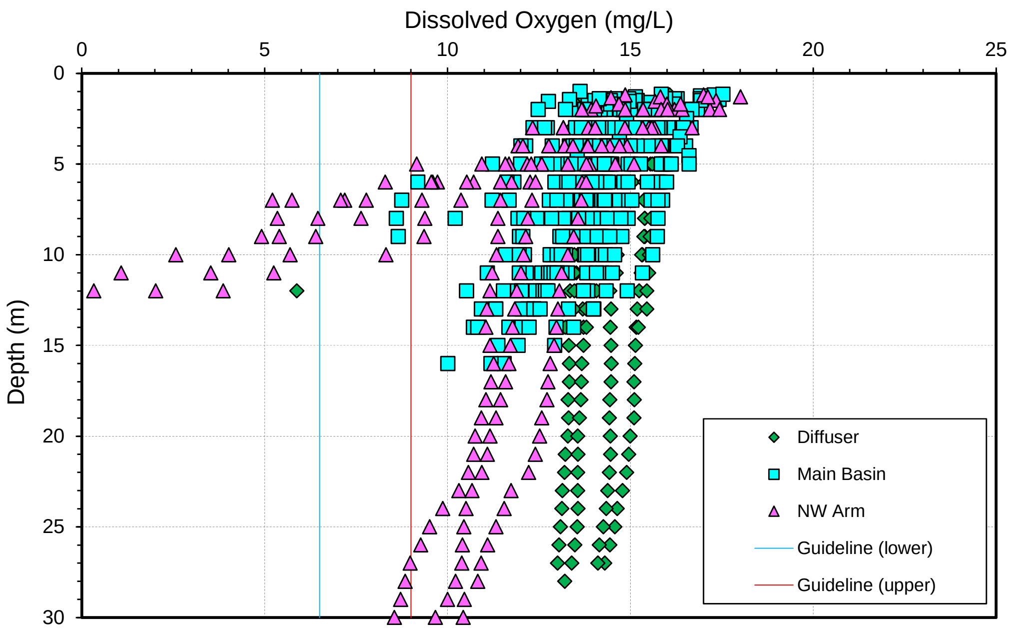


Figure 3 22 Dissolved Oxygen Concentrations during Ice-Covered Conditions, 1999 to 2012

i. 2012



j. 2013



Note: NW Arm = Northwest Arm; data shown are from representative diffuser, near-field (2004 to 2012), mid-field (2004 to 2012), far-field (2004 to 2012), main basin (2013) and NW arm stations in Snap Lake; WQG (upper) = 9.5 mg/L for early life stages; WQG (lower) = 6.5 mg/L for other life stages (CCME 1999).

mg/L= milligrams per litre; m = metre.

3.4.4.3 Summary of Key Question 3

In 2013, the following parameters displayed strong correlations to conductivity, and concentrations continued to increase above the Snap Lake normal range (i.e., baseline mean ± two standard deviations) and/or the reference lakes (Northeast Lake and Lake 13) concentrations in at least one area of Snap Lake:

- total alkalinity, TDS, reactive silica, and total hardness;
- eight major ions (bicarbonate, calcium, chloride, fluoride, magnesium, potassium, sodium, and sulphate);
- five nitrogen parameters (TN, ammonia, nitrate, nitrate and nitrite, and nitrite); and,
- eight metals (barium, boron, lithium, molybdenum, nickel, rubidium, strontium, and uranium).

Other parameters that are increasing in Snap Lake, but are not strongly correlated with conductivity, are laboratory and field pH, TKN, and manganese. Phosphorus concentrations are not increasing in Snap Lake based on both a visual review of temporal plots and statistical trend analyses.

Whole-lake averages in 2013 were slightly above predicted values for barium and uranium, likely due to model uncertainties. The 2013 whole-lake average for antimony was well above the predicted value. Since Snap Lake concentrations of antimony have not increased and are similar to reference lakes, the difference in observed and predicted values is likely related to contamination (i.e., over-estimates) in reported concentrations of antimony in Snap Lake.

The 2013 whole-lake average for parameters that were above generic AEMP benchmarks (i.e., chloride, fluoride, and nitrate) were below the EAR predictions and the upper range of the 2013 predictions. Concentrations of TDS, which is a key indicator of Mine-related changes to water quality in Snap Lake, were also below the EAR predictions and the upper range of the 2013 predictions

Minimum field pH values, which were below the AEMP benchmark, could not be compared to model predictions because pH was not modelled in the EAR or in the 2013 modelling. However, the observed increasing trends in both field and laboratory pH values indicates that the low pH values are not due to a decreasing trend in pH.

In 2013, increases in surface and bottom water DO concentrations were measured during ice-covered conditions in the main basin of Snap Lake. The increase in bottom DO concentrations during ice-covered conditions near the diffuser may result from the release of oxygenated treated effluent from the diffuser near the lake bottom.

3.4.5 Key Question 4: Are Spatial and Seasonal Patterns in Water Quality in Snap Lake and Downstream Waterbodies Consistent with Predictions Presented in the EAR and Subsequent Modelling Predictions?

This section contains qualitative assessments of spatial (i.e., horizontal and vertical) and seasonal patterns in Snap Lake water quality for field parameters, TDS, major ions, nutrients, and metals. Where patterns existed, the potential for Mine-related causes were assessed.

Seasonal and spatial patterns in key parameters were identified through plots of average concentrations in different areas of Snap Lake (i.e., diffuser stations, main basin, and northwest arm) and in the reference lakes (i.e., Northeast Lake and Lake 13) by season (i.e., open-water and ice-cover). Data collected from Northeast Lake and Lake 13 are presented for comparison to help separate natural variability and background environmental changes from potential effects and patterns resulting from the Mine. Seasonal and spatial plots for all laboratory parameters are presented in Appendix 3H.

Water quality data from the furthest downstream AEMP station, KING01, were reviewed to identify potential changes in water quality at a location 25 km downstream of Snap Lake. Refer to Section 11.3 for information collected in the lakes immediately downstream of Snap Lake as part of the Downstream Lakes Special Study; a short summary is provided in this section.

3.4.5.1 Spatial Patterns and Seasonal Variation for Snap Lake

Field Parameters

Conductivity

Conductivity is a reliable field indicator of TDS, as illustrated by the close relationship between conductivity and TDS in Snap Lake from 2004 to 2013 in Figure 3-23. In 2013, field conductivity in Snap Lake has increased from previous years, ranging from 116 microSiemens per centimetre (µS/cm) in the northwest arm in July to 659 µS/cm at the diffuser station SNP 02-20f in May (De Beers 2013b; Appendix 3C, Table 3C-1).

Spatial variability in conductivity in Snap Lake in 2013 was consistent with recent years in that field conductivity measurements in the main basin and northwest arm were related to proximity to the discharge and hydraulic connectivity (Figures 3-24 to 3-32; De Beers 2010, 2011, 2012b). The order of measured conductivity from highest to lowest was: diffuser stations; main basin beyond the diffuser stations; and, northwest arm (Figure 3-24). The conductivity values at the diffuser stations and in the rest of the main basin were relatively similar; differences in conductivity within the main basin of Snap Lake are becoming less distinguishable (Figures 3-24 to 3-32).

The greatest spatial variability in conductivity within Snap Lake was observed in the northwest arm (Figures 3-24 to 3-31). In May, conductivities in the main basin varied by 107 µS/cm in contrast to a variability of 249 µS/cm in conductivities in the northwest arm (Figure 3-26). In 2013, conductivity in the northwest arm was lower compared to the main basin of Snap Lake, consistent with historical spatial trends since the Mine began operating (De Beers 2005a, 2006, 2007a, 2008a, 2009, 2010, 2011, 2012b, 2013b). The highest conductivity in the northwest arm occurred at SNAP23 and SNAP29, which are closest to the connection between the northwest arm and the main basin (Figures 3-25 to Figures 3-28). The influence of the treated effluent discharged in the main basin of Snap Lake was predicted to affect the northwest arm of Snap Lake at a slower rate than the main basin, due to the limited hydraulic connectivity between the northwest arm and the main basin (De Beers 2002).

Vertical profiles of conductivity varied in the main basin with distance from the diffuser during the late ice-covered season (Figure 3-26). In May, field conductivity at all diffuser stations and at SNAP03, which is the closest station to the diffuser stations in the main basin, increased from the surface to a depth of about 5 to 10 m, and then remained relatively consistent to the bottom of the lake. Conductivity at stations further from the diffuser stations (i.e., SNAP06, SNAP09, and SNAP11A) increased from the surface to a depth of about 5 to 10 m, and then decreased to the bottom of the lake. Conductivity at SNAP08, located at the outlet of Snap Lake, was higher at the surface and then decreased with depth to the bottom of the lake during ice-covered conditions (Figure 3-26).

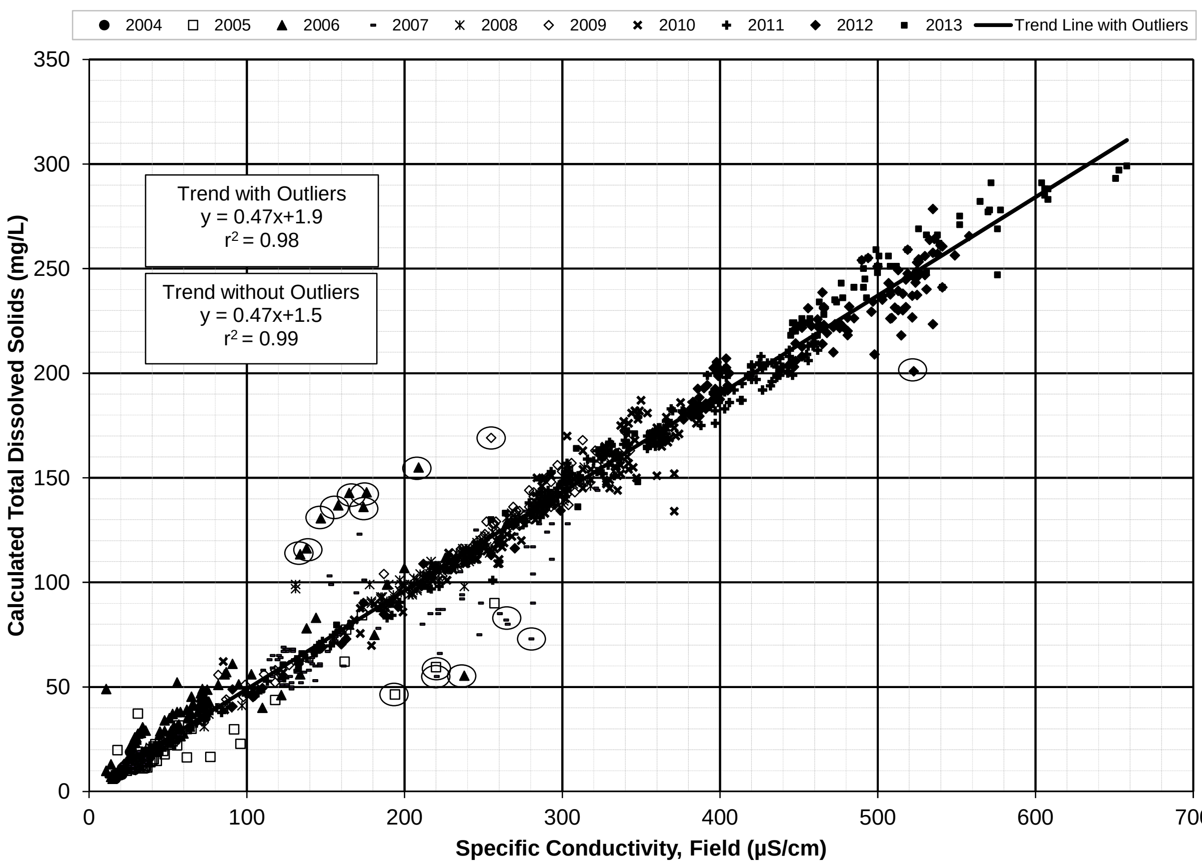
Higher conductivity at mid-depth in the main basin may be due to the influence from the diffuser, which has ports that discharge treated effluent away from the bottom of the lake. At the start of Mine operations, the higher density of the treated effluent plume relative to the lake water caused the treated effluent plume to settle back down to the bottom. However, as TDS concentrations in the lake have increased, the difference between the density of the plume and the lake water has decreased, and in 2013 the plume continued to be situated mid-column rather than sinking to the bottom as observed prior to 2009.

Vertical profiles also show that the treated effluent is situated near the bottom of the water column in the northwest arm, particularly at stations SNAP23 and SNAP29 (Figures 3-25 to 3-28). The density difference between the high conductivity water from the main basin and the lower conductivity water in the northwest arm may cause the high conductivity water to settle to the bottom as it enters the northwest arm.

Open-water profiles of conductivity showed that the water column is mostly mixed at the start of the open-water season in July (Figure 3-27), and becomes fully mixed at most of the stations in Snap Lake near the end of the open-water season in September (Figure 3-28). Conductivity increased with depth at the deepest diffuser station (SNP 02-20e), at two deep main basin stations (SNAP09 and SNAP11A), and at SNAP23 and SNAP29 in the northwest arm in July, indicating the water column was not yet completely mixed (Figure 3-27). Complete vertical mixing had occurred at most stations in the main basin by September; no vertical conductivity gradients were observed in Snap Lake, with the exception of SNAP29 in the northwest arm. The vertical conductivity gradient at SNAP29 was likely related to denser high conductivity water from the main basin sinking as it entered the lower density and conductivity waters of the northwest arm during the open-water season (Figures 3-28 and 3-32). The lack of a discernible vertical gradient at most of the sampling stations in Snap Lake, near the end of the open-water season is consistent with increased mixing of the treated effluent due to wind-driven currents.

In 2013, conductivity measured in Northeast Lake and Lake 13 during ice-covered and open-water conditions was lower compared to measured conductivity throughout Snap Lake (Figures 3-24 to 3-28). A slight vertical gradient in conductivity at the surface water layer was evident in Northeast Lake in February and May and in Lake 13 in May during ice-covered conditions; complete vertical mixing occurred during open-water conditions. The small surface layer gradient observed during ice-covered conditions is likely due to the exclusion of naturally occurring salts as the surface freezes (Pieters and Lawrence 2009). This observation is common in northern lakes, but is only noticeable at a few stations in Snap Lake (e.g., SNAP20B and SNAP02A in February [Figure 3-25]) due to the elevated dissolved solids concentrations associated with treated effluent.

Figure 3-23 Relationship between Field Conductivity and Total Dissolved Solids in Snap Lake, 2004 to 2013

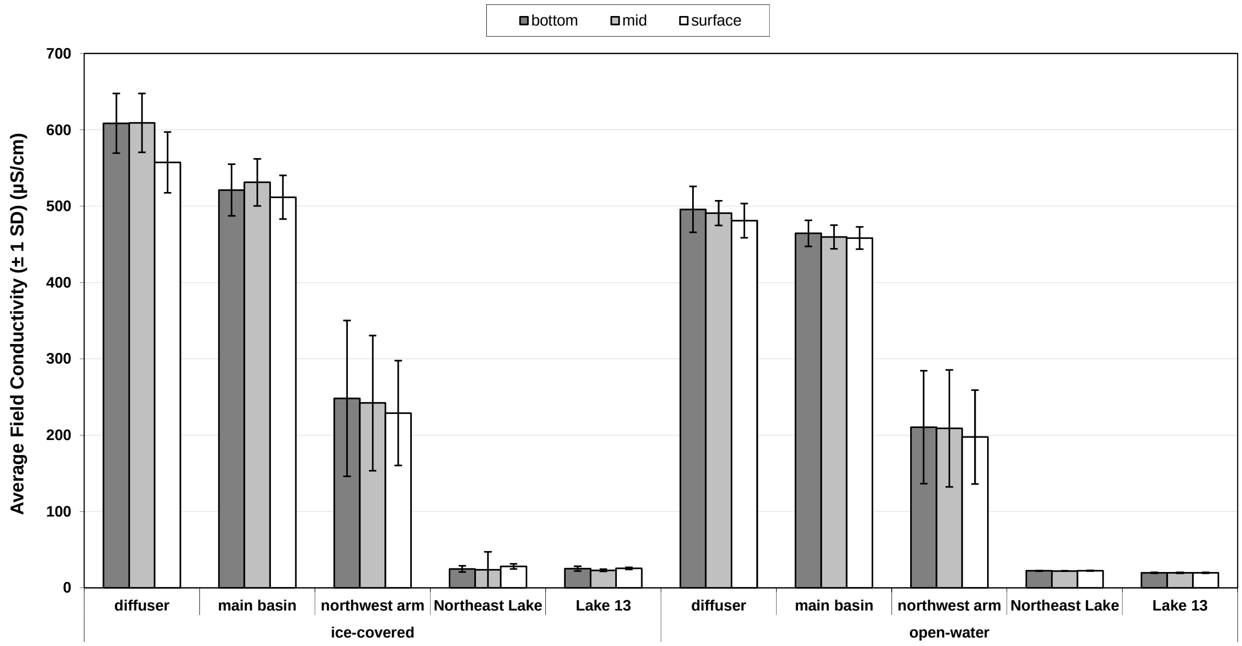


Note: Circled data points are outliers.

Total dissolved solids calculated using formula adapted from Method 1030 E in Standard Methods for the Examination of Water and Wastewater, 21st Edition (APHA 2005).

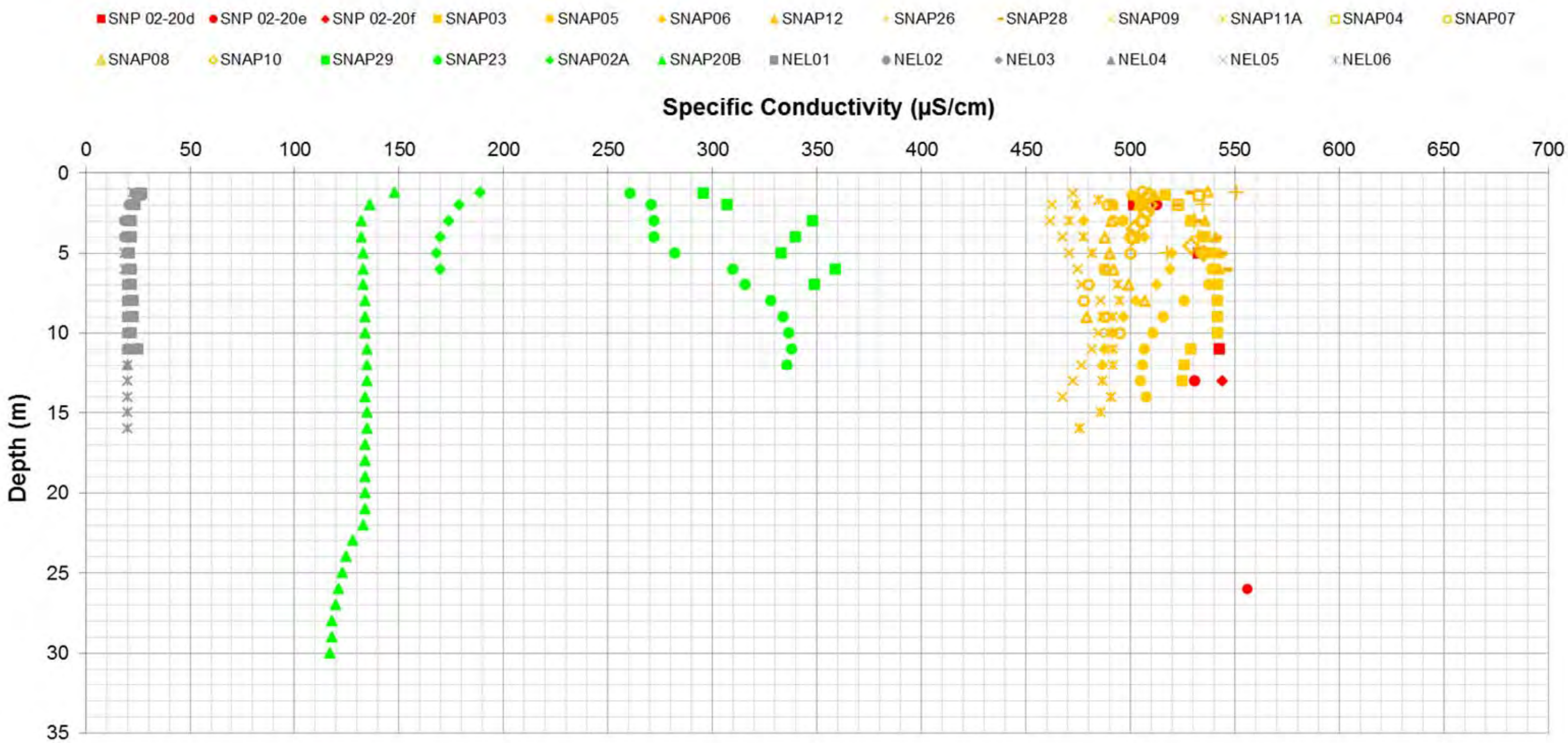
mg/L = milligrams per litre; µS/cm = microSiemens per centimetre.

Figure 3-24 Average of Field Conductivity (± 1 SD) in Snap Lake, Northeast Lake, and Lake 13, 2013



SD = standard deviation; µS/cm = microSiemens per centimetre.

Figure 3-25 Field and Laboratory Conductivity Profiles in Snap Lake and Northeast Lake, February 2013



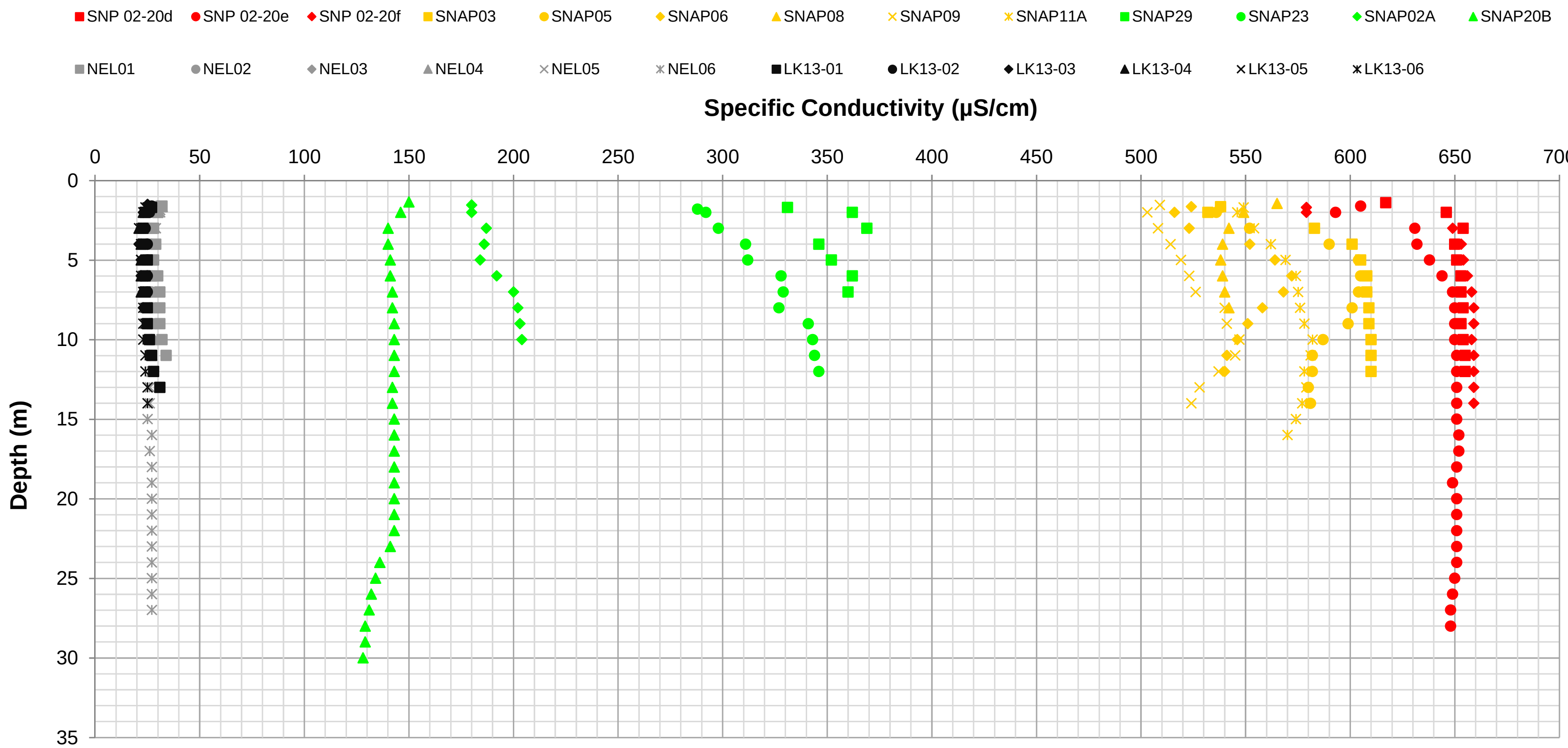
Note: Grey symbols represent Northeast Lake stations; green symbols represent the northwest arm stations; light blue, dark blue and yellow symbols represent the main basin stations; and, red symbols represent the diffuser stations.

Field conductivity collected at diffuser stations were considered potential erroneous due to the calibration error during February sampling program. Therefore, laboratory conductivity data were included for the diffuser stations in the plot. Details are included in Appendix 3A.

Due to a field error the February profile was collected approximately 240 m away from the standard sampling location NEL06. Water quality data collected at the non-standard station were similar to water quality data at other locations in Northeast Lake in February; therefore the data were considered valid and included in the assessment.

m = metre; µS/cm = microSiemens per centimetre.

Figure 3-26 Field Conductivity Profiles in Snap Lake, Northeast Lake and Lake 13, May 2013

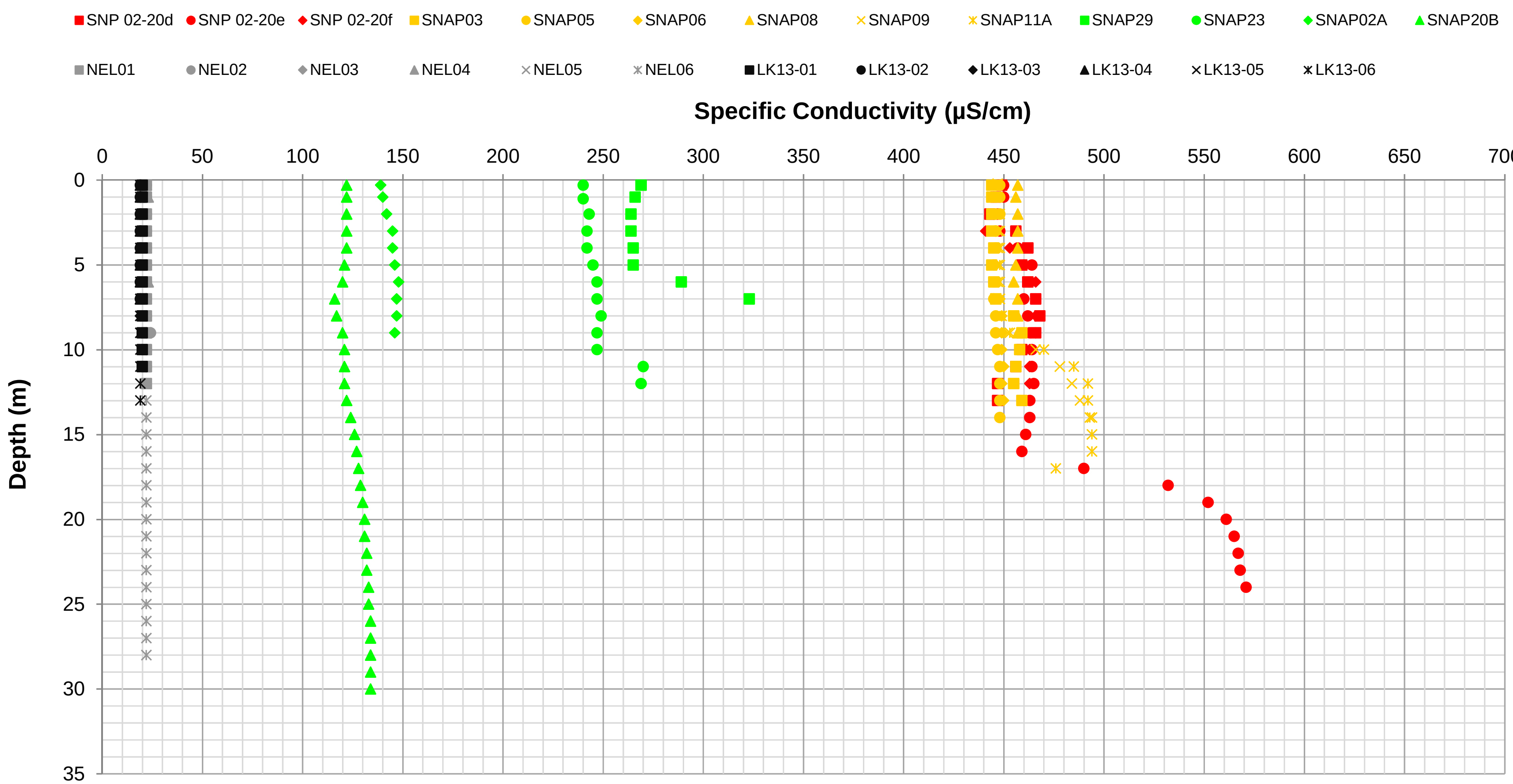


Note: Grey symbols represent Northeast Lake stations; black symbols represent Lake 13 stations; green symbols represent the northwest arm stations; yellow symbols represent the main basin stations; and, red symbols represent the diffuser stations.

Due to a field error the May and July water samples and field profiles were collected approximately 250 to 500 m away from the standard sampling locations LK13-03, LK13-05, and LK13-06. Water quality data collected at these non-standard locations in May and July were similar to water quality data collected at the other locations in Lake 13 in May and July respectively; therefore, the data were considered valid and included in the assessment.

m = metre; µS/cm = microSiemens per centimetre.

Figure 3-27 Field Conductivity Profiles Snap Lake, Northeast Lake and Lake 13, July 2013

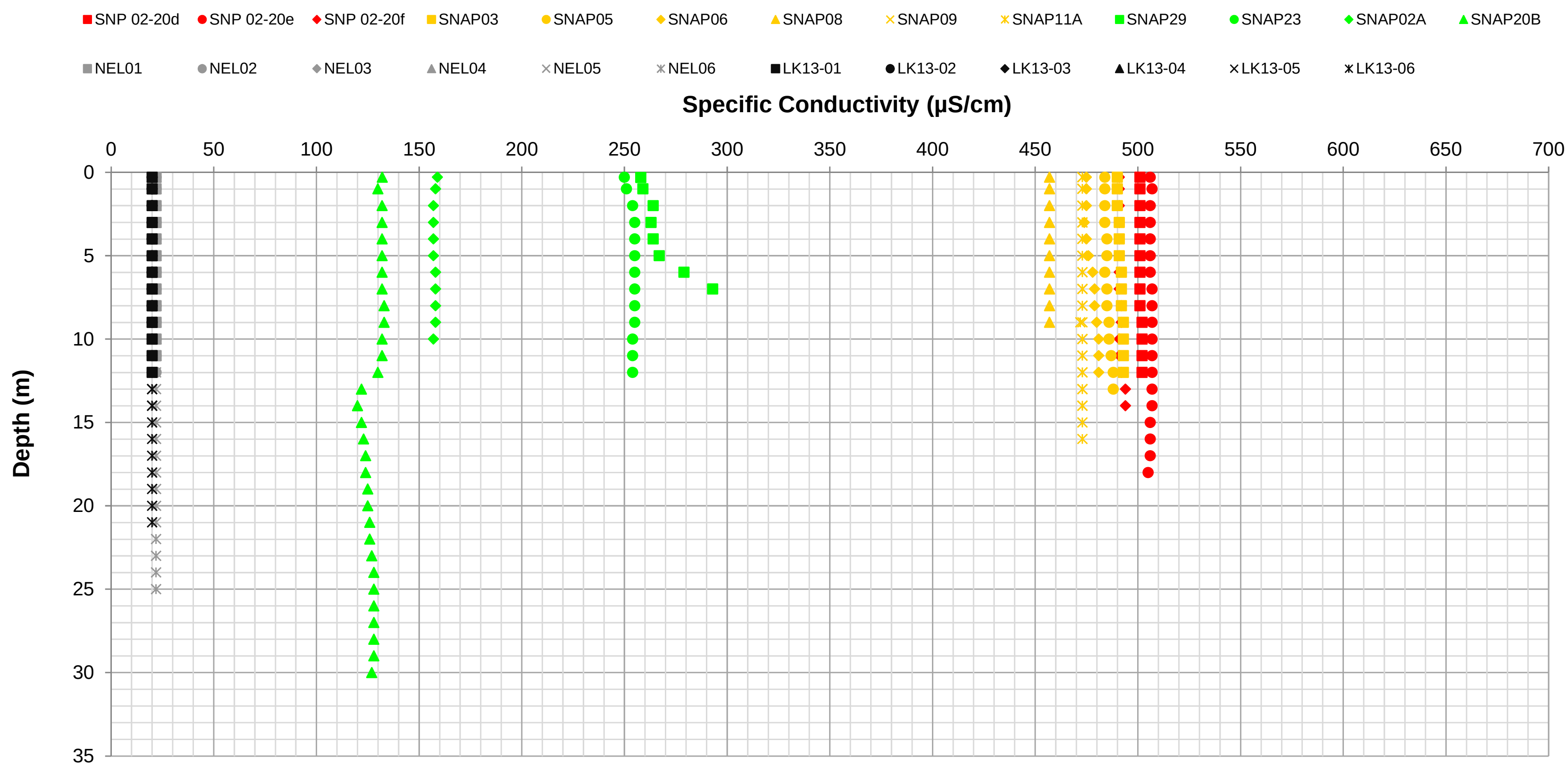


Note: Grey symbols represent Northeast Lake stations; black symbols represent Lake 13 stations; green symbols represent the northwest arm stations; yellow symbols represent the main basin stations; and, red symbols represent the diffuser stations.

Due to a field error the May and July water samples and field profiles were collected approximately 250 to 500 m away from the standard sampling locations LK13-03, LK13-05, and LK13-06. Water quality data collected at these non-standard locations in May and July were similar to water quality data collected at the other locations in Lake 13 in May and July respectively; therefore, the data were considered valid and included in the assessment.

m = metre; µS/cm = microSiemens per centimetre.

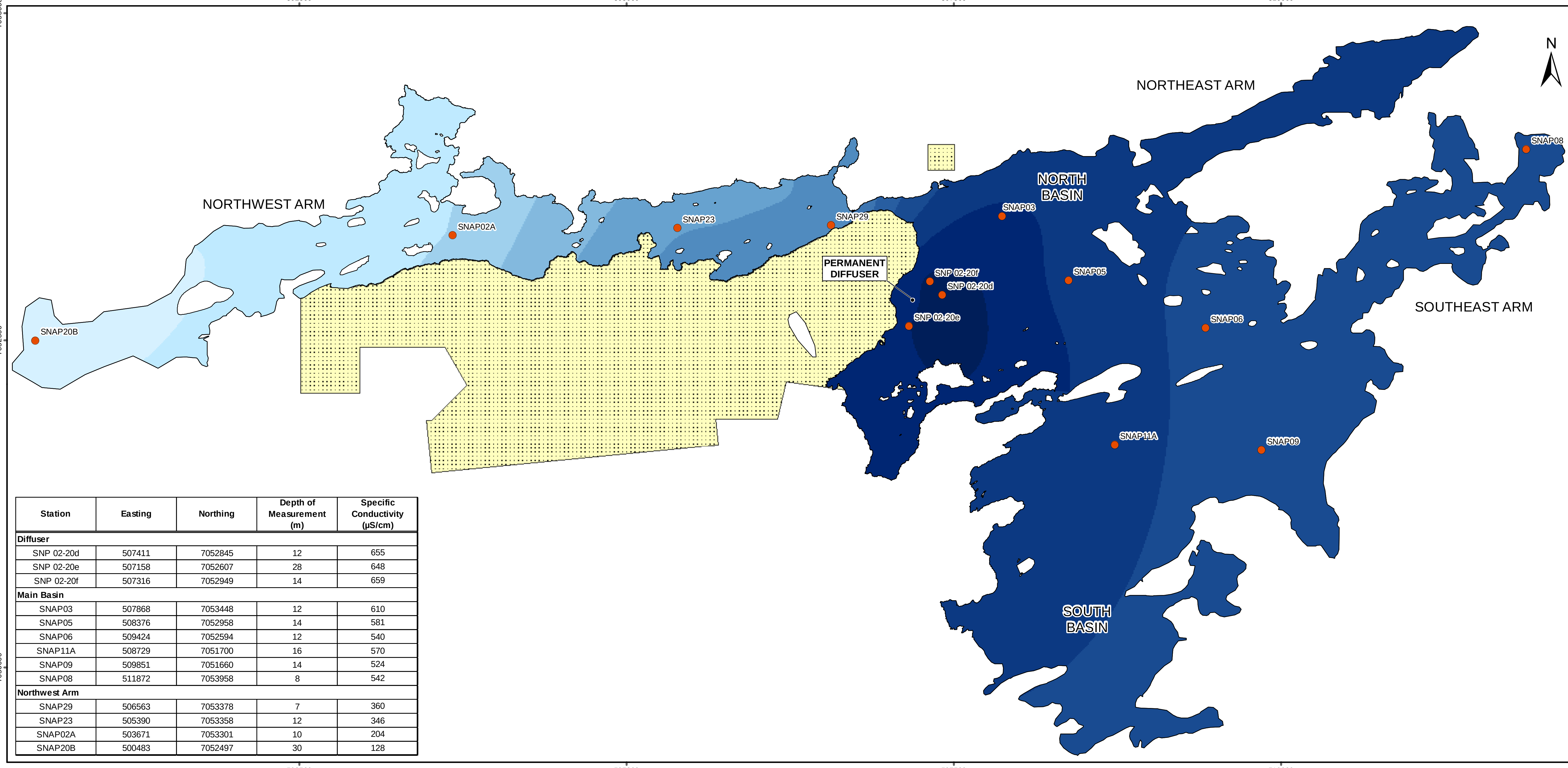
Figure 3-28 Field Conductivity Profiles Snap Lake, Northeast Lake and Lake 13, September 2013



Note: Grey symbols represent Northeast Lake stations; black symbols represent Lake 13 stations; green symbols represent the northwest arm stations; yellow symbols represent the main basin stations; and, red coloured symbols represent the diffuser stations.

Due to a field error, the September and October water samples and profiles were collected 70 m west of the regular SNP 02-20e monitoring location. Water near the diffuser is well-mixed, as evidenced by the consistent water quality data among the three locations presented in the plot. Therefore, the data collected 70 m west of SNP 02-20e were considered valid and representative of SNP 02-20e.

m = metre; µS/cm = microSiemens per centimetre.



Station	Easting	Northing	Depth of Measurement (m)	Specific Conductivity (µS/cm)
Diffuser				
SNP 02-20d	507411	7052845	12	655
SNP 02-20e	507158	7052607	28	648
SNP 02-20f	507316	7052949	14	659
Main Basin				
SNAP03	507868	7053448	12	610
SNAP05	508376	7052958	14	581
SNAP06	509424	7052594	12	540
SNAP11A	508729	7051700	16	570
SNAP09	509851	7051660	14	524
SNAP08	511872	7053958	8	542
Northwest Arm				
SNAP29	506563	7053378	7	360
SNAP23	505390	7053358	12	346
SNAP02A	503671	7053301	10	204
SNAP20B	500483	7052497	30	128

LEGEND

- Aquatic Effects Monitoring Program Station
- Snap Lake Mine Footprint

Specific Conductivity

- < 150 µS/cm
- 150 - 200 µS/cm
- 200 - 250 µS/cm
- 250 - 300 µS/cm
- 300 - 350 µS/cm
- 350 - 400 µS/cm
- 400 - 450 µS/cm
- 450 - 500 µS/cm
- 500 - 550 µS/cm
- 550 - 600 µS/cm
- 600 - 650 µS/cm
- > 650 µS/cm

REFERENCES

Base data source: CanVec

NOTES

< = less than
> = greater than
µS/cm = microSiemens per centimetre
m = metres

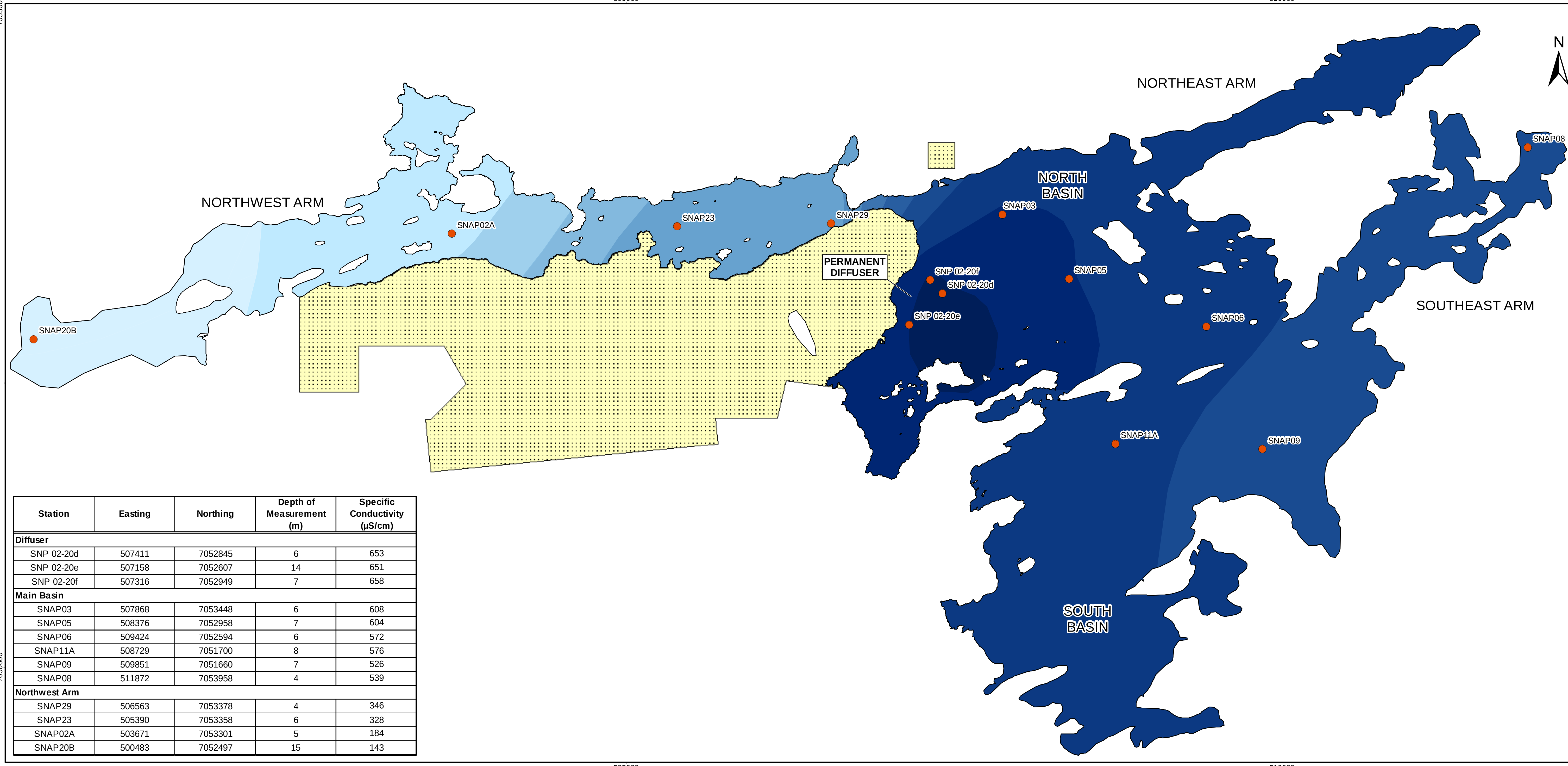
SNAP LAKE MINE

Bottom Conductivity in Snap Lake, May 2013

PROJECTION: UTM Zone 12	DATUM: NAD83
500 0 500 SCALE 1:30,000 METRES	DE BEERS Group of Companies
FILE No: 1413490004-7001_3_29	DATE: April 9, 2014
JOB NO: 14-1349-0004	REVISION NO: A
OFFICE: GOLD-CAL	DRAWN: JC / CHECK: PMC

Figure 3-29

I:\CLIENTS\DE_BEERS\Snap_Lake\Mapping\MXD\AEMP\2013\Fig3-29_AEMP_Bottom_Conductivity_SnapLake_May2013_FINAL.mxd



Station	Easting	Northing	Depth of Measurement (m)	Specific Conductivity (µS/cm)
Diffuser				
SNP 02-20d	507411	7052845	6	653
SNP 02-20e	507158	7052607	14	651
SNP 02-20f	507316	7052949	7	658
Main Basin				
SNAP03	507868	7053448	6	608
SNAP05	508376	7052958	7	604
SNAP06	509424	7052594	6	572
SNAP11A	508729	7051700	8	576
SNAP09	509851	7051660	7	526
SNAP08	511872	7053958	4	539
Northwest Arm				
SNAP29	506563	7053378	4	346
SNAP23	505390	7053358	6	328
SNAP02A	503671	7053301	5	184
SNAP20B	500483	7052497	15	143

LEGEND

- Aquatic Effects Monitoring Program Station
- Snap Lake Mine Footprint

Specific Conductivity

- < 150 µS/cm
- 150 - 200 µS/cm
- 200 - 250 µS/cm
- 250 - 300 µS/cm
- 300 - 350 µS/cm
- 350 - 400 µS/cm
- 400 - 450 µS/cm
- 450 - 500 µS/cm
- 500 - 550 µS/cm
- 550 - 600 µS/cm
- 600 - 650 µS/cm
- > 650 µS/cm

REFERENCES

Base data source: CanVec

NOTES

< = less than
> = greater than
µS/cm = microSiemens per centimetre
m = metres

SNAP LAKE MINE

Mid-Depth Conductivity in Snap Lake, May 2013

PROJECTION: UTM Zone 12
DATUM: NAD83

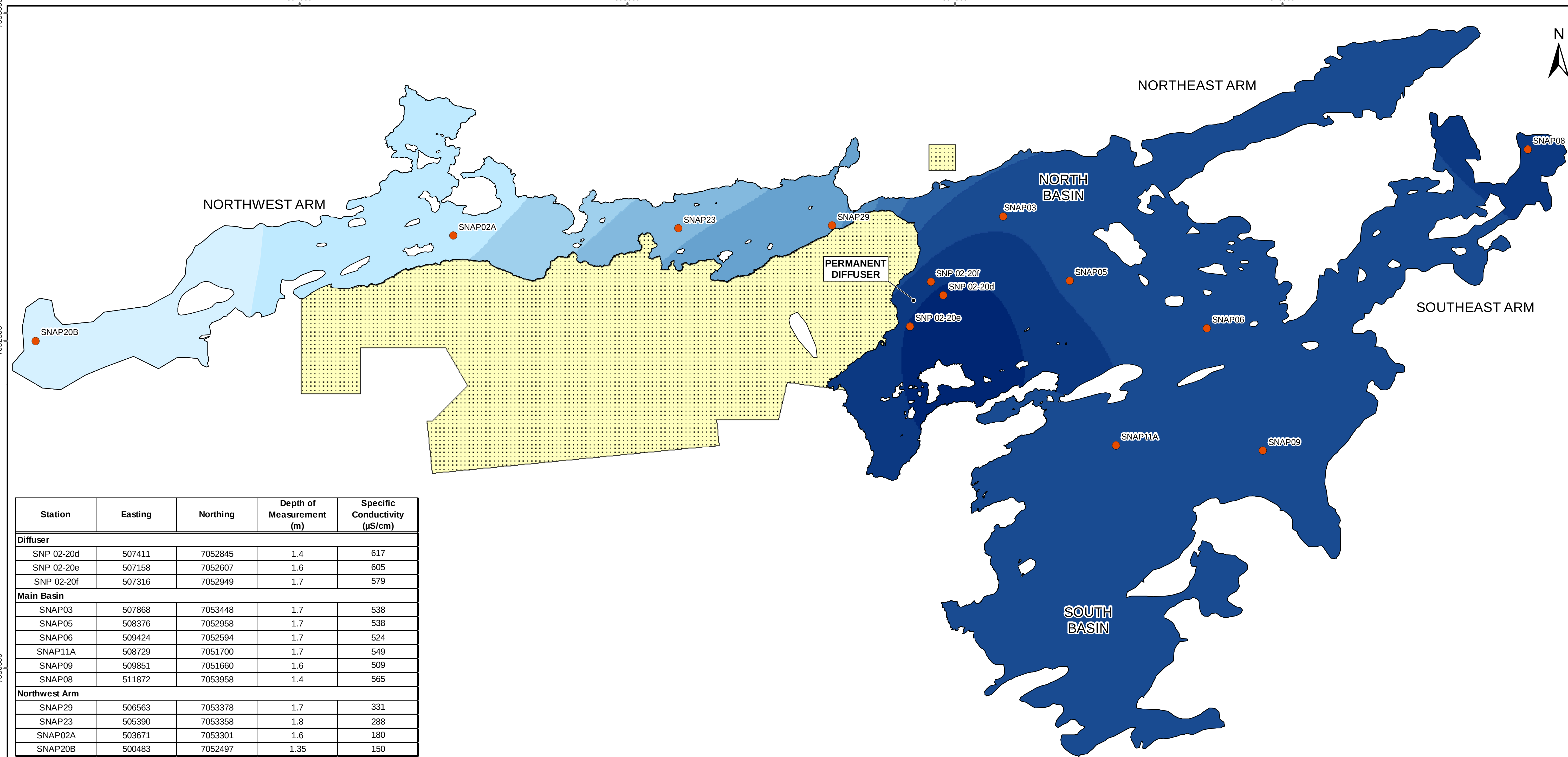
500 0 500
SCALE 1:30,000 METRES

DE BEERS
GROUP OF COMPANIES

FILE No: 1413490004-7001_3_30
DATE: April 9, 2014
JOB NO: 14-1349-0004
REVISION NO: A
OFFICE: GOLD-CAL
DRAWN: JC
CHECK: PMC

Figure 3-30

I:\CLIENTS\DE_BEERS\Snap_Lake\Mapping\MXD\AEMP2013\Fig3-30_AEMP_Mid-Depth_Conductivity_SnapLake_May2013_FINAL.mxd



Station	Easting	Northing	Depth of Measurement (m)	Specific Conductivity (µS/cm)
Diffuser				
SNP 02-20d	507411	7052845	1.4	617
SNP 02-20e	507158	7052607	1.6	605
SNP 02-20f	507316	7052949	1.7	579
Main Basin				
SNAP03	507868	7053448	1.7	538
SNAP05	508376	7052958	1.7	538
SNAP06	509424	7052594	1.7	524
SNAP11A	508729	7051700	1.7	549
SNAP09	509851	7051660	1.6	509
SNAP08	511872	7053958	1.4	565
Northwest Arm				
SNAP29	506563	7053378	1.7	331
SNAP23	505390	7053358	1.8	288
SNAP02A	503671	7053301	1.6	180
SNAP20B	500483	7052497	1.35	150

LEGEND

Aquatic Effects Monitoring Program Station

Snap Lake Mine Footprint

Specific Conductivity

- < 150 µS/cm
- 150 - 200 µS/cm
- 200 - 250 µS/cm
- 250 - 300 µS/cm
- 300 - 350 µS/cm
- 350 - 400 µS/cm
- 400 - 450 µS/cm
- 450 - 500 µS/cm
- 500 - 550 µS/cm
- 550 - 600 µS/cm
- 600 - 650 µS/cm

REFERENCES

Base data source: CanVec

NOTES

< = less than
> = greater than
µS/cm = microSiemens per centimetre
m = metres

SNAP LAKE MINE

Surface Conductivity in Snap Lake, May 2013

PROJECTION: UTM Zone 12 | DATUM: NAD83

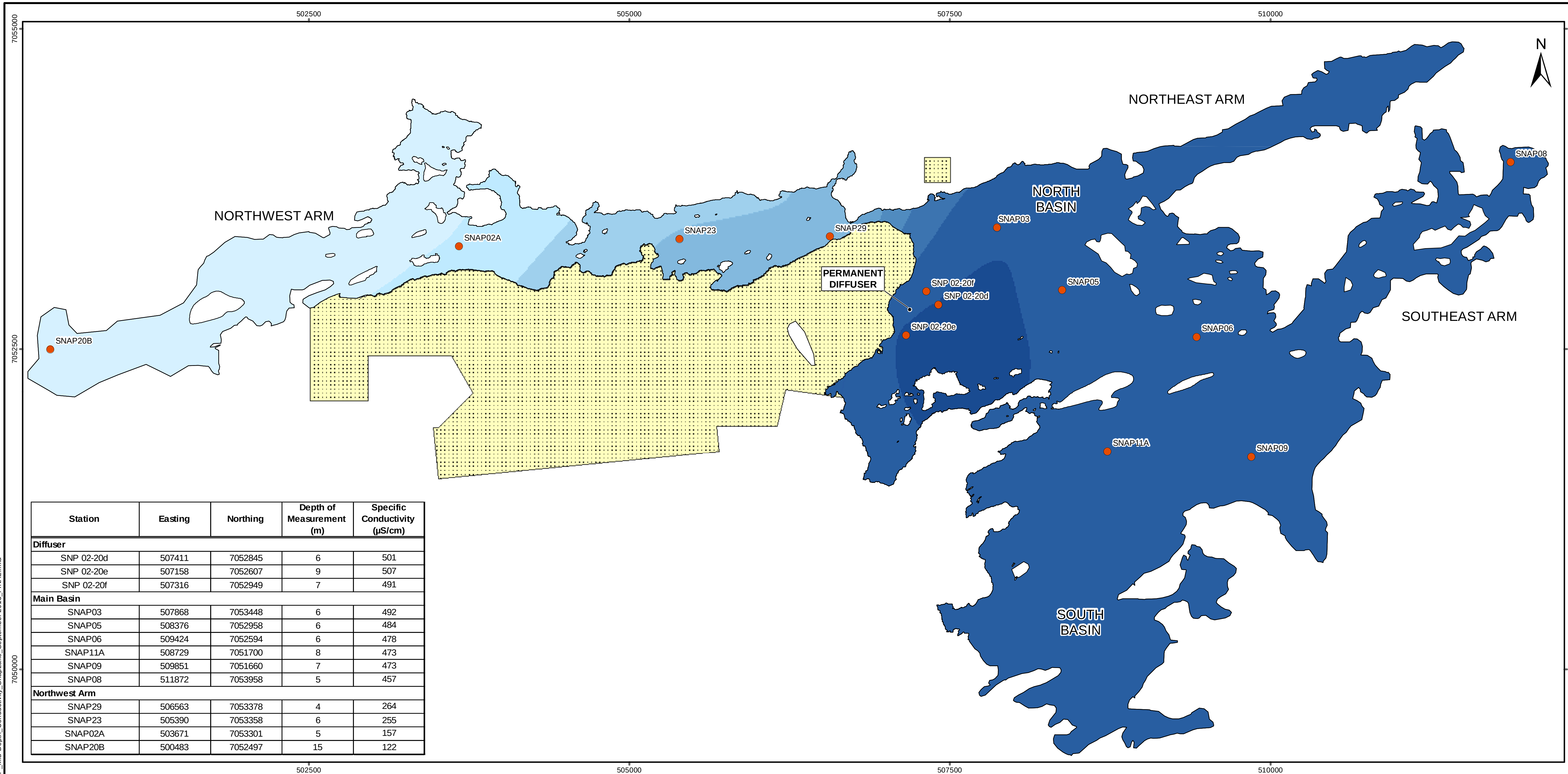
500 0 500
SCALE 1:30,000 METRES

DE BEERS
GROUP OF COMPANIES

FILE No: 1413490004-7001_3_31
DATE: April 9, 2014
JOB NO: 14-1349-0004
REVISION NO: A
OFFICE: GOLD-CAL
DRAWN: JC
CHECK: PMC

Figure 3-31

I:\CLIENTS\DE_BEERS\Snap_Lake\Mapping\MXD\AEMP2013\Fig3-31_AEMP_Surface_Conductivity_May 2013_FINAL.mxd



Station	Easting	Northing	Depth of Measurement (m)	Specific Conductivity (µS/cm)
Diffuser				
SNP 02-20d	507411	7052845	6	501
SNP 02-20e	507158	7052607	9	507
SNP 02-20f	507316	7052949	7	491
Main Basin				
SNAP03	507868	7053448	6	492
SNAP05	508376	7052958	6	484
SNAP06	509424	7052594	6	478
SNAP11A	508729	7051700	8	473
SNAP09	509851	7051660	7	473
SNAP08	511872	7053958	5	457
Northwest Arm				
SNAP29	506563	7053378	4	264
SNAP23	505390	7053358	6	255
SNAP02A	503671	7053301	5	157
SNAP20B	500483	7052497	15	122

I:\CLIENTS\DE_BEERS\Snap_Lake\Mapping\MXD\AEMP2013\Fig3-32_AEMP_Mid-Depth_Conductivity_SnapLake_September 2013_FINAL.mxd

LEGEND

- Aquatic Effects Monitoring Program Station
- Snap Lake Mine Footprint

Specific Conductivity

- < 150 µS/cm
- 150 - 200 µS/cm
- 200 - 250 µS/cm
- 250 - 300 µS/cm
- 300 - 350 µS/cm
- 350 - 400 µS/cm
- 400 - 450 µS/cm
- 450 - 500 µS/cm
- 500 - 550 µS/cm

REFERENCES

Base data source: CanVec

NOTES

< = less than
> = greater than
µS/cm = microSiemens per centimetre
m = metres

SNAP LAKE MINE

Mid-Depth Conductivity in Snap Lake, September 2013

PROJECTION: UTM Zone 12
DATUM: NAD83

500 0 500
SCALE 1:30,000 METRES

DE BEERS
GROUP OF COMPANIES

FILE No: 1413490004-7001_3_32
DATE: April 9, 2014
JOB NO: 14-1349-0004
REVISION NO: A
OFFICE: GOLD-CAL
DRAWN: JC
CHECK: PMC

Figure 3-32

Dissolved Oxygen

Concentrations of DO in Snap Lake varied by season and with water depth, ranging from 0.3 mg/L in May at SNAP23 to 18 mg/L in February 2013 at SNAP29 in the northwest arm; (Appendix 3B, Table 3B-1).

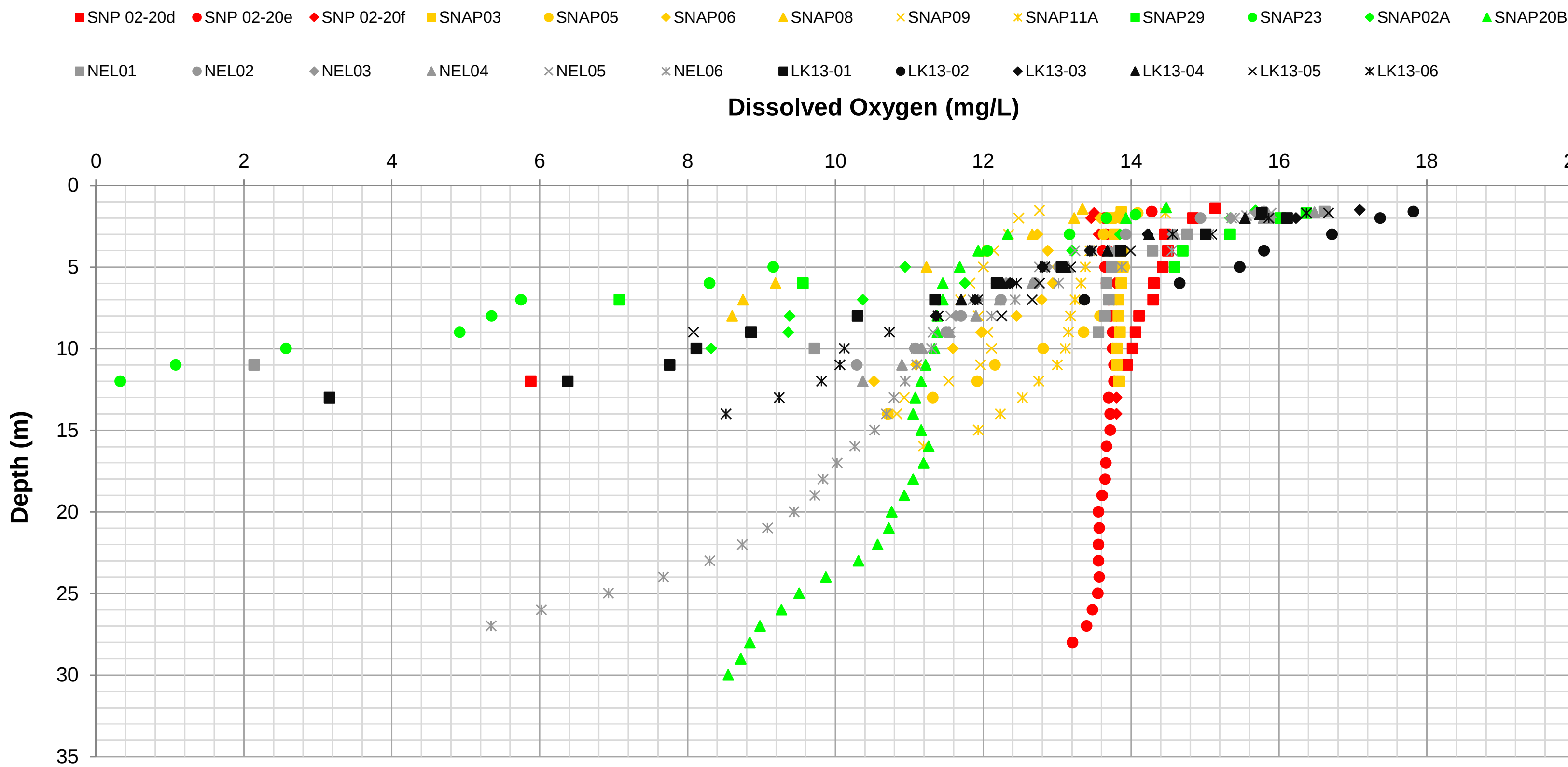
During ice-covered conditions, vertical gradients in DO occurred at most monitoring stations in Snap Lake and the reference lakes; lower DO concentrations occurred near the bottom of the lake (Figures 3-33 and Figure 3-7, Panels a to k). A lack of re-aeration potential due to ice-cover and oxygen consumption through natural biological and chemical processes in the water column can cause naturally low bottom DO concentrations in lakes during winter conditions (Catalan et al. 2002). Less of a vertical gradient was observed at the diffuser stations (SNP 02-20d, SNP 02-20e, and SNP 02-20f) during ice-covered conditions, where the DO concentrations were relatively similar throughout the water column (Figure 3-33 and Figure 3-7, Panels i and k). The treated effluent, which is well-oxygenated, may be increasing naturally low concentrations of DO at the diffuser stations during ice-covered conditions.

Maximum DO concentrations during open-water conditions were lower than maximum DO concentrations during ice-covered conditions in Snap Lake and the reference lakes (Figure 3-34 and Figure 3-7; Panels a to k), which is also consistent with natural variations in DO. As the temperature of lake water increases during the open-water season, the saturation point of DO decreases; therefore, the water has a lower capacity for DO.

Vertical gradients in DO were not evident during open-water conditions in Snap Lake, with the exception of the deepest station in the northwest arm, SNAP20B, in July and September (Figure 3-7, Panel b), and the deepest station in the main basin, the diffuser station SNP 02-20e, in August (Figure 3-7, Panel j). The gradients of decreasing concentrations of DO occurred at deeper depths at SNP 02-20e compared to SNAP20B, likely related to a combination of the diffuser discharging well-oxygenated treated effluent near SNP 02-20e (i.e., adding oxygen to the water column, increasing naturally lower DO concentrations) and potentially higher wind-driven mixing in the main-basin relative to the northwest arm

Vertical gradients of decreasing DO with increasing depth were observed during ice-covered conditions at Northeast Lake and Lake 13 stations. During open-water conditions, the DO concentrations in Northeast Lake and Lake 13 remained consistent through the water column, with the exception of stations NEL01 and NEL06, in Northeast Lake in July (Figure 3-7, panels l and m). The bottom DO concentration measured at station NEL01 was 1.5 mg/L lower and was likely caused by oxygen consumption through natural biological and chemical processes near the bottom sediments. The high DO concentration measured at the bottom of NEL06 was likely related to the decrease in the water temperature at the same depth; as temperature decreases the saturation point of DO increases, allowing a higher capacity for DO in the water.

Figure 3-33 Dissolved Oxygen Profiles in Snap Lake, Northeast Lake and Lake 13, May 2013

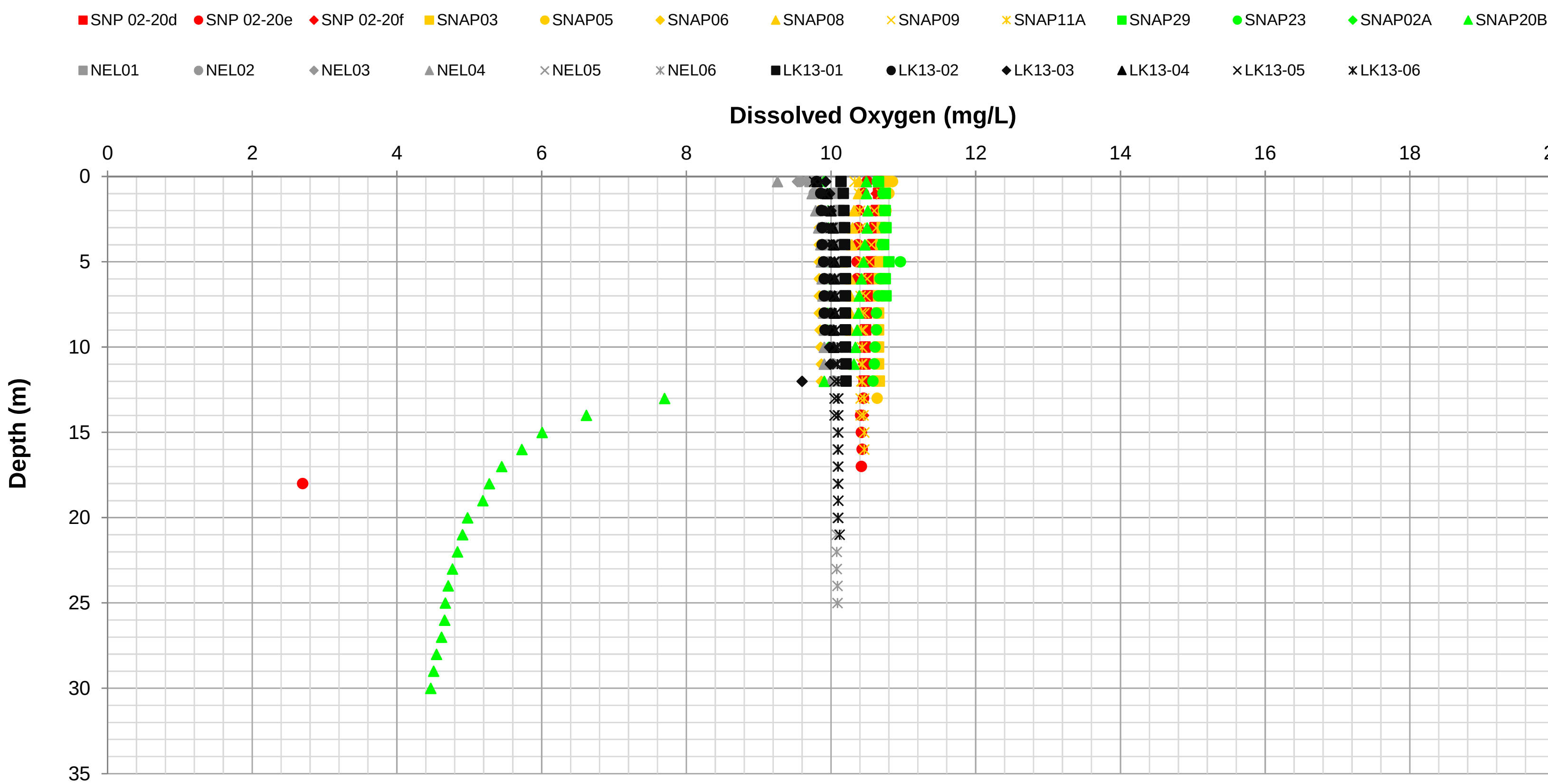


Note: Grey symbols represent Northeast Lake stations; black symbols represent Lake 13 stations; green symbols represent the northwest arm stations; yellow symbols represent the main basin stations; and, red symbols represent the diffuser stations.

Due to a field error the May and July water samples and field profiles were collected approximately 250 to 500 m away from the standard sampling locations LK13-03, LK13-05, and LK13-06. Water quality data collected at these non-standard locations in May and July were similar to water quality data collected at the other locations in Lake 13 in May and July respectively; therefore, the data were considered valid and included in the assessment.

m = metre; mg/L = milligrams per litre.

Figure 3-34 Dissolved Oxygen Profiles in Snap Lake, Northeast Lake and Lake 13, September 2013



Note: Grey symbols represent Northeast Lake stations; black symbols represent Lake 13 stations; green symbols represent the northwest arm stations; yellow symbols represent the main basin stations; and, red symbols represent the diffuser stations.

Due to a field error, the September and October water samples and profiles were collected 70 m west of the regular SNP 02-20e monitoring location. Water near the diffuser is well-mixed, as evidenced by the consistent water quality data among the three locations presented in the plot. Therefore, the data collected 70 m west of SNP 02-20e were considered valid and representative of SNP 02-20e.

m = metre; mg/L = milligrams per litre.

pH

In 2013, the range in Snap Lake field pH values was 5.3 to 8.1, a similar range to the field pH range measured in previous years (De Beers 2005a, 2006, 2007a, 2008a, 2009, 2010, 2011, 2012b, 2013b). Treated effluent has elevated pH and alkalinity due to the high neutralization potential of kimberlite and elevated TDS and hardness. The pH of the treated effluent is also adjusted and maintained during the treatment process.

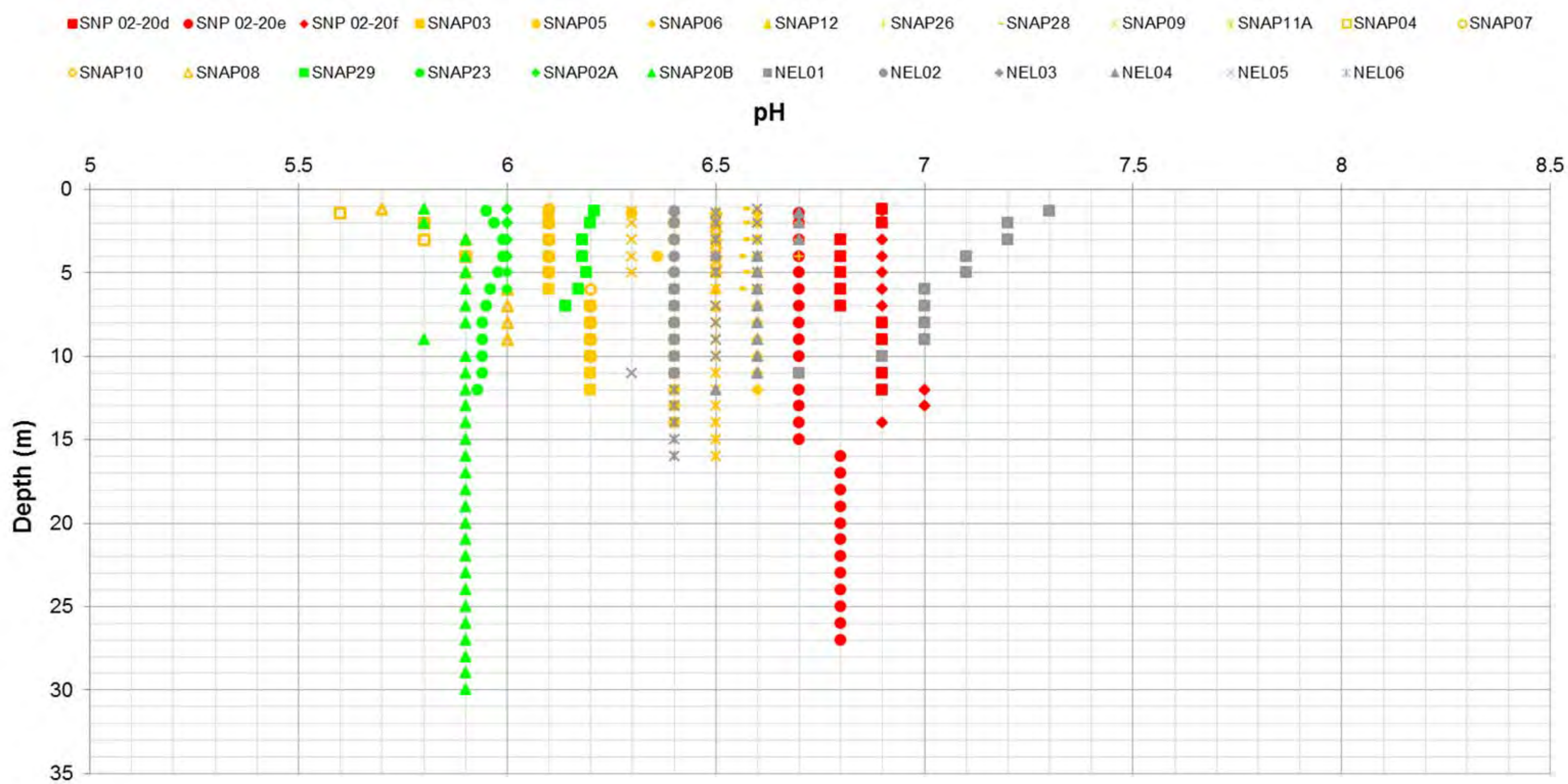
During ice-covered and open-water conditions, the pH values were slightly higher at the stations closest to the diffuser stations (i.e., at SNAP03, SNAP05, and SNAP11A) and decreased with distance away from the diffuser (Figures 3-35 to 3-38). The field pH data collected at most of the stations in May were considered potential erroneous due a calibration error; therefore, the closest available field pH data to May (i.e., March field pHs) were used to represent field pH patterns for ice-covered conditions (Figure 3-36).

Vertical gradients in pH in Snap Lake were only observed at two shallow stations farther away from the diffuser (SNAP04 and SNAP08) during ice-covered conditions; pH values were relatively similar throughout the water column at all other stations in Snap Lake (Figures 3-35 and 3-36). The lack of vertical gradients in 2013 pH values differs from the 2012 results, which showed vertical gradients in pH values during ice-covered conditions (De Beers 2013b).

During open-water conditions, pH values were consistent throughout the water column at the shallow stations; deeper stations (SNP 02-20e and SNAP20B) showed a decreasing gradient in pH (Figure 3-38). This decrease may be caused by deep water processes and water column-sediment interactions, such as respiration, sediment decomposition, and redox reactions, as well as less influence from wind-driven mixing. The vertical spatial patterns observed at SNP 02-20e and SNAP20B in September 2013 were consistent with previous years (De Beers 2012b, 2013b).

During ice-covered conditions, the pH values in Northeast Lake were similar to the levels measured at Snap Lake (Figures 3-35 and 3-36). Similar to Snap Lake, pH gradients were not observed in Northeast Lake with the exception of NEL01, where pH decreased with depth. During open-water conditions, the pH values in Northeast Lake and Lake 13 were slightly lower than the pH values measured in Snap Lake (Figures 3-37 and 3-38), consistent with laboratory measurements of pH (Appendix 3H). Vertical gradients in pH were not observed in Northeast Lake or Lake 13, with the exception of three locations (LK13-01, NEL01, and NEL04), where pH increased with depth (Figures 3-37 and 3-38).

Figure 3-35 pH Profiles in Snap Lake and Northeast Lake, February 2013

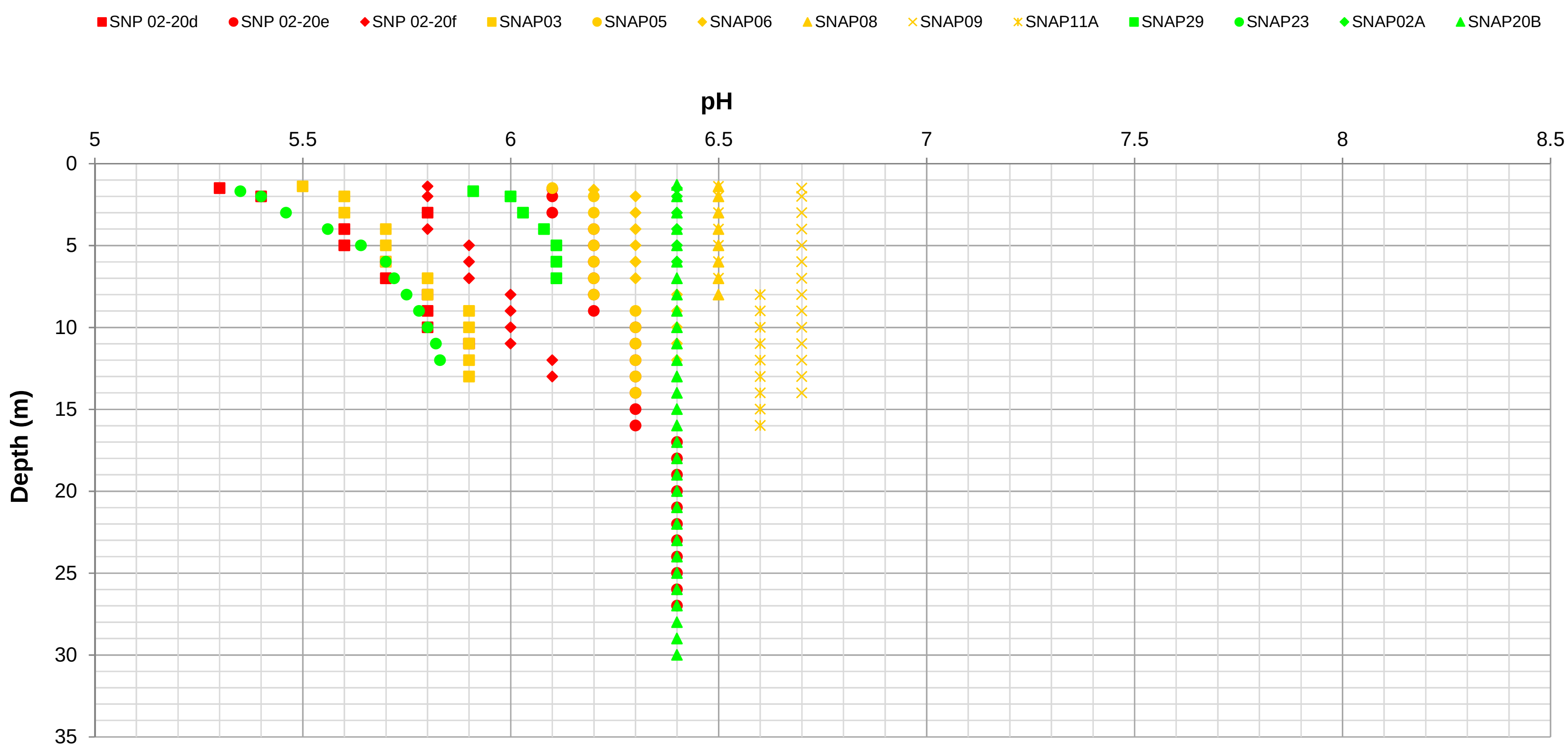


Note: Grey symbols represent Northeast Lake stations; green symbols represent the northwest arm stations; light blue, dark blue and yellow symbols represent the main basin stations; and, red symbols represent the diffuser stations.

Due to a field error the February profile was collected approximately 240 m away from the standard sampling location NEL06. Water quality data collected at the non-standard station were similar to water quality data at other locations in Northeast Lake in February; therefore the data were considered valid and included in the assessment

m = metre.

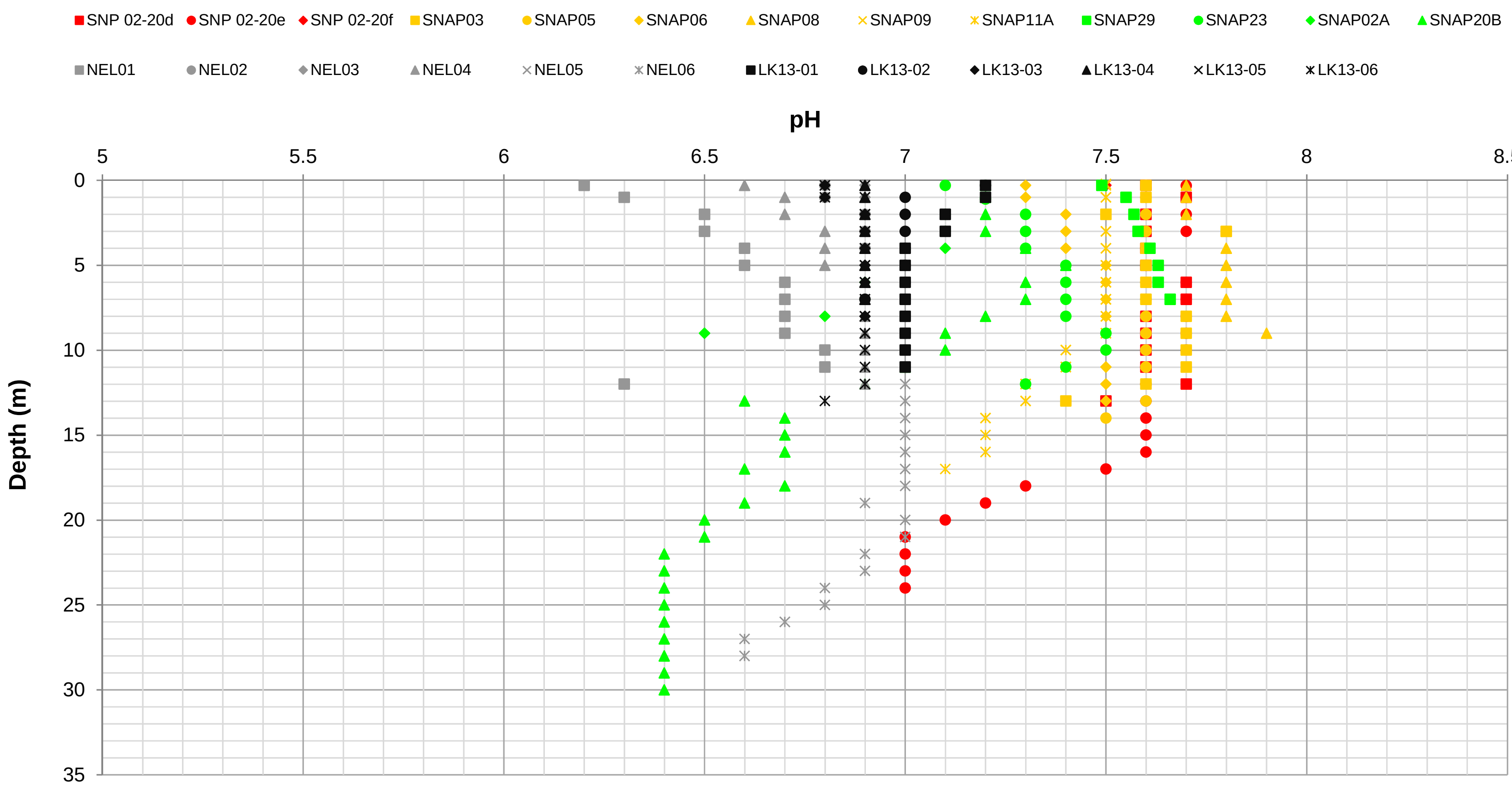
Figure 3-36 pH Profiles in Snap Lake and Northeast Lake, March 2013



Note: Green symbols represent the northwest arm stations; yellow symbols represent the main basin; and, red symbols represent the diffuser stations. Field pH data was not collected at Northeast Lake and Lake 13.

m = metre.

Figure 3-37 pH Profiles in Snap Lake, Northeast Lake and Lake 13, July 2013

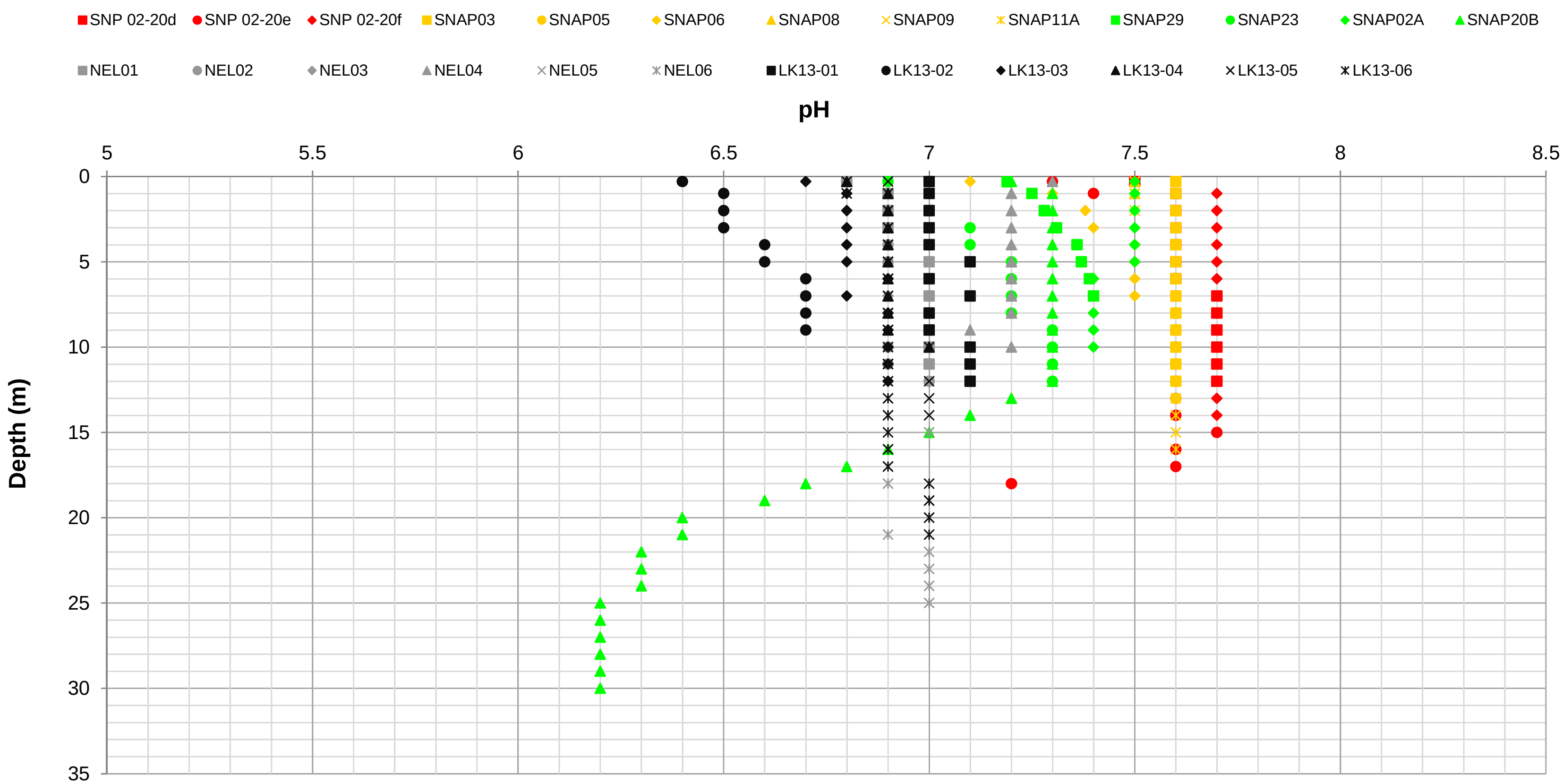


Note: Grey symbols represent Northeast Lake stations; black symbols represent Lake 13 stations; green symbols represent the northwest arm stations; yellow symbols represent the main basin stations; and, red symbols represent the diffuser stations.

Due to a field error the May and July water samples and field profiles were collected approximately 250 to 500 m away from the standard sampling locations LK13-03, LK13-05, and LK13-06. Water quality data collected at these non-standard locations in May and July were similar to water quality data collected at the other locations in Lake 13 in May and July respectively; therefore, the data were considered valid and included in the assessment.

m = metre.

Figure 3-38 pH Profiles in Snap Lake, Northeast Lake and Lake 13, September 2013



Note: Grey symbols represent Northeast Lake stations; black symbols represent Lake 13 stations; green symbols represent the northwest arm stations; yellow symbols represent the main basin stations; and, red symbols represent the diffuser stations.

Due to a field error, the September and October water samples and profiles were collected 70 m west of the regular SNP 02-20e monitoring location. Water near the diffuser is well-mixed, as evidenced by the consistent water quality data among the three locations presented in the plot. Therefore, the data collected 70 m west of SNP 02-20e were considered valid and representative of SNP 02-20e.

m = metre.

Water Temperature

In 2013, surface water temperatures in Snap Lake varied from 0.1 degrees Celsius (°C) during ice-covered conditions in March at SNAP04 to 19.4°C during open-water conditions in July at SNP 02-20f (Appendix 3C). The maximum temperature increased slightly (i.e., higher by 1.3°C) compared to 2012 (De Beers 2013b). Consistent with 2006 to 2012, vertical temperature gradients were observed at:

- most stations during the ice-covered conditions (Figures 3-39 and 3-40);
- most stations deeper than 5 m during early open-water conditions (July, Figure 3-41); and,
- one of the deepest stations, SNAP20B, near the end of the open-water season (September, Figure 3-42).

During ice-covered conditions, temperatures at all stations increased with depth; however, temperature increases were observed to a lesser degree at the diffuser stations (Figures 3-39 to 3-42). During ice-covered conditions, the water column profiles at the diffuser stations and at the stations close to the diffuser stations in the main basin were cooler than the water column profiles at the other locations in the lake. The temperature of the effluent in the WTP (SNP 02-17B) in winter was approximately 7°C warmer than the lake during ice-covered conditions (Figure 3-43). The cooler temperatures near the diffuser were likely due to heat loss at the open-water area in the ice above the diffuser structure.

During early open-water conditions in July, the shallow surface depths, including the euphotic zone of the lake, warmed, so temperatures at all the stations gradually decreased with depth, with the exception of four deep stations (Figure 3-41). At the deepest northwest arm station (SNAP20B), two deeper stations in the main basin (SNAP09 and SNAP11A), and the deepest diffuser stations (SNP 02-20e), temperatures decreased gradually (approximately 0.5°C per m) until approximately 7 m, 12 m, and 17 m, respectively, at which point temperatures dropped 4°C or more within approximately a 2 m depth. The thermograph data from the two deep water locations in Snap Lake (i.e., SNP 02-20e and SNAP20B) showed a similarly sharp decline in temperature at these depths in July (Section 2.4.4). The large decrease in temperature within a small change in depth indicates the presence of a thermocline at these locations during early open-water conditions; such thermoclines can hinder mixing. The lack of mixing was evident in the conductivity measurements at these stations below the relevant thermocline depths, most prominently at the diffuser (SNP 02-20e) and main basin stations (SNAP09 and SNAP11A) because of their proximity to the diffuser.

When the water column was mixed in September, observed temperatures were relatively uniform throughout the water column, except at the deepest station in the northwest arm, SNAP20B and at the deep diffuser station (SNP 02-20e). The temperatures at SNAP20B remained cooler near the bottom of the lake but the thermocline occurred at a deeper depth (13 m from surface), indicating that a greater top portion of the water column was well-mixed compared to July at this deep location. The difference in mixing was also evident in the conductivity profile, where a small increase in conductivity (10 µS/cm) was observed at the thermocline depth. The thermocline at SNAP20B in September was also evident in the thermograph data (Section 2.4.4). The thermographs at SNP 02-20e also showed a thermocline in

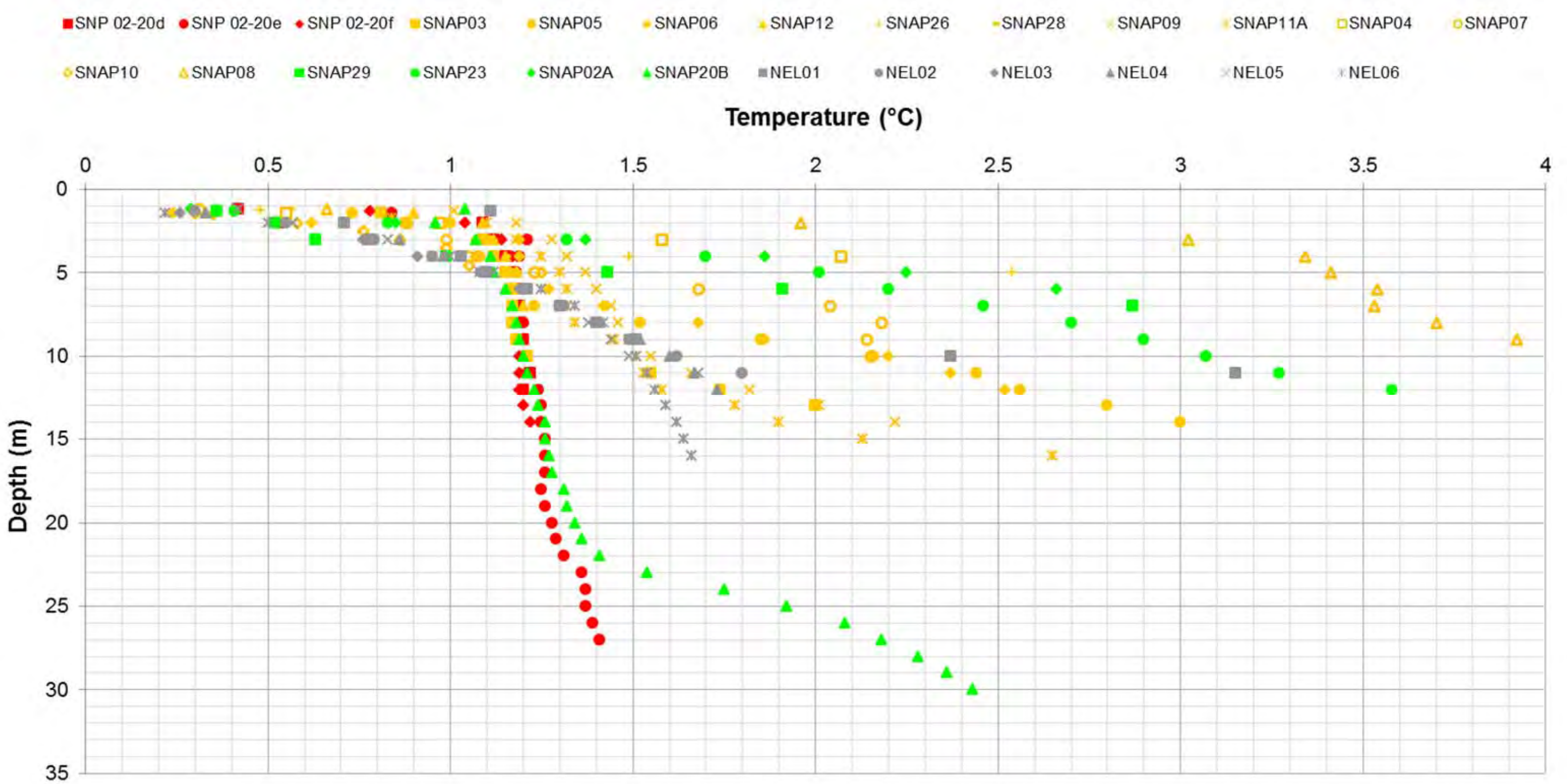
September at 29 m (Section 2.4.4 and Figure 3-42); the potential increase in conductivity at the thermocline at SNP 02-20e could not be assessed because water profiles were mistakenly collected from a shallower location near SNP 02-20e.

Temperatures in Northeast Lake and Lake 13 increased with depth and were similar to the temperatures in the main basin of Snap Lake during the ice-covered conditions (Figures 3-39 and 3-40). During the early open-water conditions in July, the temperatures in Northeast Lake and Lake 13 were lower than the temperatures measured in Snap Lake.

Temperatures in the Northeast Lake and Lake 13 were uniform throughout the water column under open-water conditions, with the exception of the temperature at the deepest station in Northeast Lake (NEL06) and a deeper station (i.e., 18 m) in Lake 13 (LK13-03) in July (Figure 3-42). A thermocline, where temperatures decreased sharply with depth, was observed at approximately 22 m at NEL06 in July but, by September, no gradient was observed in the temperature profile at NEL06. The results of the thermographs used to assess water temperatures at deep locations in Northeast Lake indicated the presence of a thermocline at approximately 24 m in September (Section 2.4.4); this thermocline may not have been observed in the water quality profile reported for NEL06 because the profiles were only reported to a depth of 25 m at NEL06 in September (Section 3.2.1.1).

During the late open-water conditions in September, temperatures in Northeast Lake and Lake 13 were slightly lower than temperatures measured at Snap Lake stations, with the exception of the outlet station (SNAP08; Figure 3-42). Temperatures have historically been higher in Northeast Lake compared to Snap Lake during the late open-water period (De Beers 2008a, 2009, 2010, 2012b, 2013b), with the exception of temperatures measured in 2010 (De Beers 2011).

Figure 3-39 Water Temperature Profiles in Snap Lake and Northeast Lake, February 2013

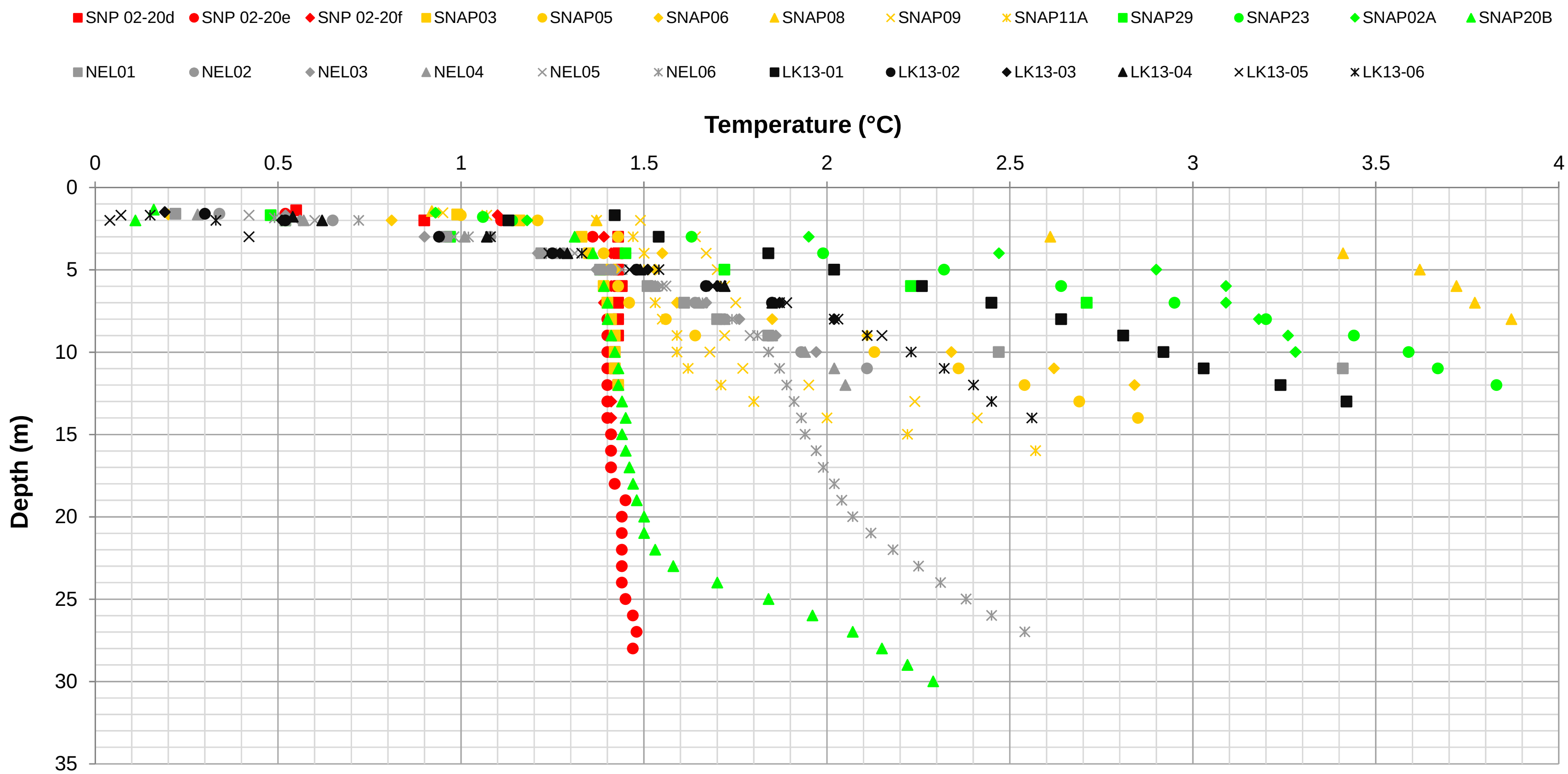


Note: Grey symbols represent Northeast Lake stations; green symbols represent the northwest arm stations; light blue symbols represent the mid-field stations; dark blue symbols represent the far-field stations; yellow symbols represent the near-field stations; and, red symbols represent the diffuser stations.

Due to a field error the February profile was collected approximately 240 m away from the standard sampling location NEL06. Water quality data collected at the non-standard station were similar to water quality data at other locations in Northeast Lake in February; therefore the data were considered valid and included in the assessment

m = metre; °C = degrees Celsius.

Figure 3-40 Water Temperature Profiles in Snap Lake, Northeast Lake and Lake 13, May 2013

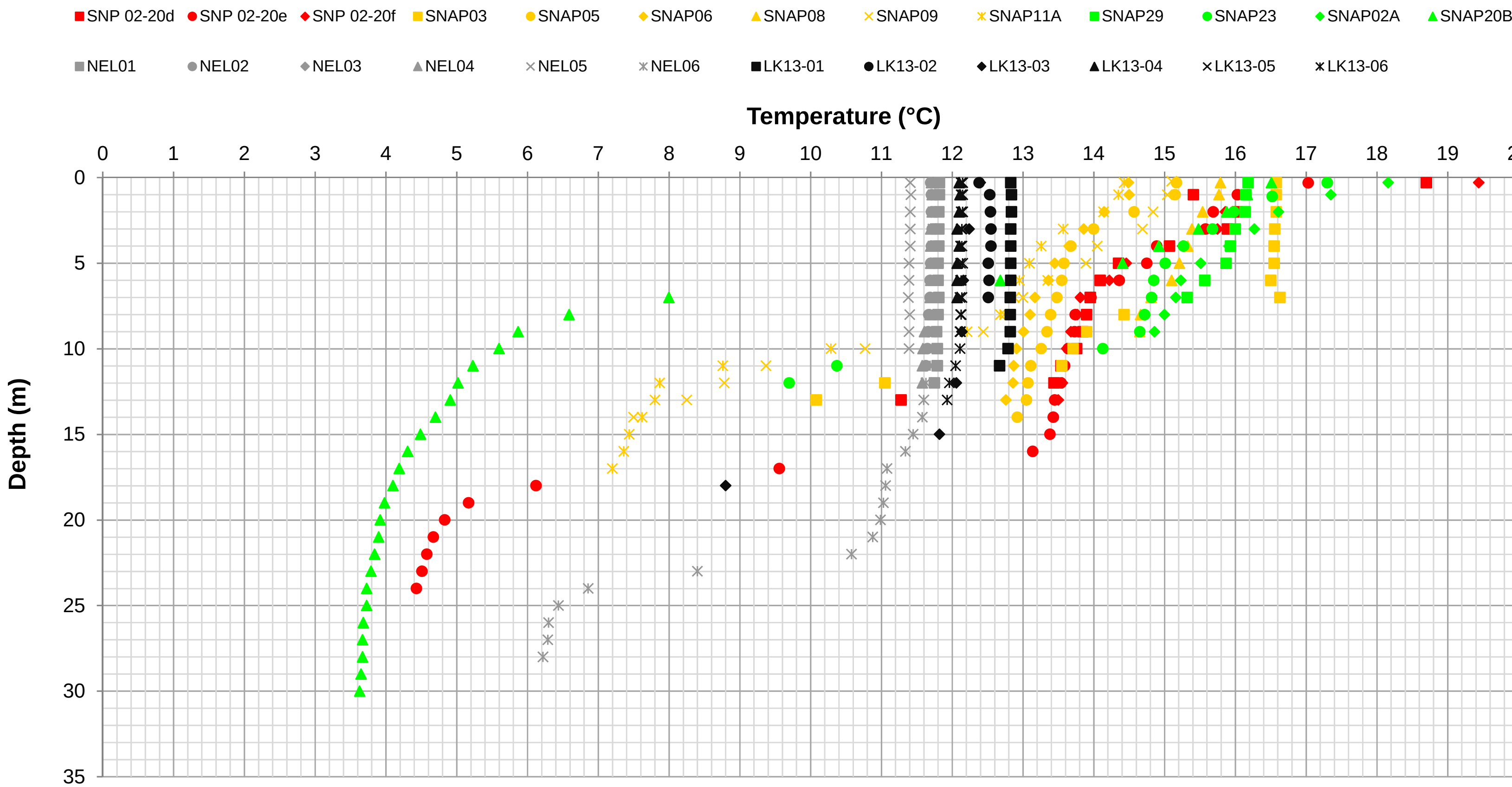


Note: Grey symbols represent Northeast Lake stations; black symbols represent Lake 13 stations; green symbols represent the northwest arm stations; yellow symbols represent the main basin stations; and, red symbols represent the diffuser stations.

Due to a field error the May and July water samples and field profiles were collected approximately 250 to 500 m away from the standard sampling locations LK13-03, LK13-05, and LK13-06. Water quality data collected at these non-standard locations in May and July were similar to water quality data collected at the other locations in Lake 13 in May and July respectively; therefore, the data were considered valid and included in the assessment.

m = metre; °C = degrees Celsius.

Figure 3-41 Water Temperature Profiles in Snap Lake, Northeast Lake and Lake 13, July 2013

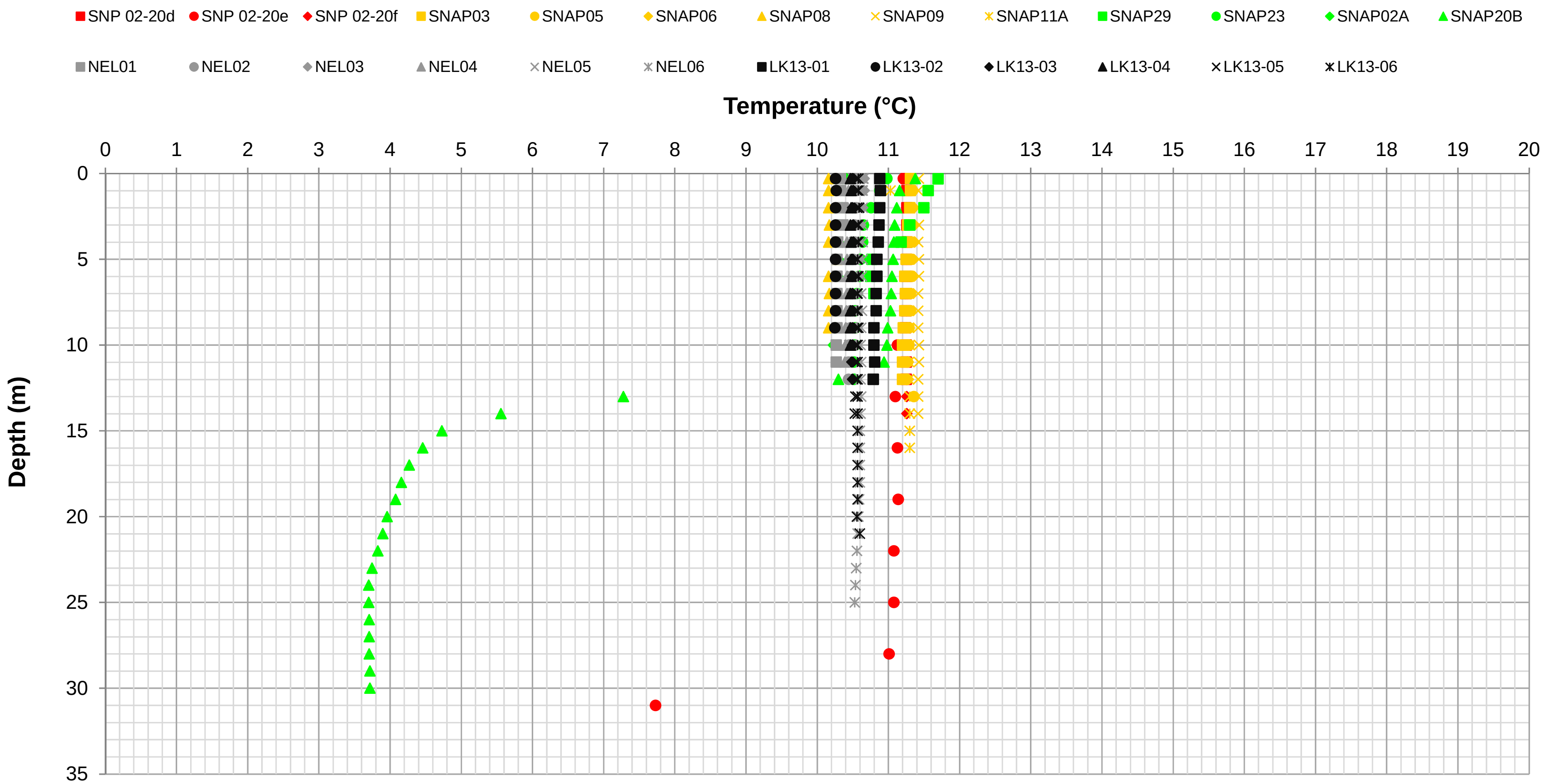


Note: Grey symbols represent Northeast Lake stations; black symbols represent Lake 13 stations; green symbols represent the northwest arm stations; yellow symbols represent the main basin stations; and, red symbols represent the diffuser stations.

Due to a field error the May and July water samples and field profiles were collected approximately 250 to 500 m away from the standard sampling locations LK13-03, LK13-05, and LK13-06. Water quality data collected at these non-standard locations in May and July were similar to water quality data collected at the other locations in Lake 13 in May and July respectively; therefore, the data were considered valid and included in the assessment. Therefore, the temperature collected at LK13-03 on July 15, 2013 to assess seasonal water temperatures (Section 2.4.4) was shown in this figure.

m = metre; °C = degrees Celsius.

Figure 3-42 Water Temperature Profiles in Snap Lake, Northeast Lake and Lake 13, September 2013

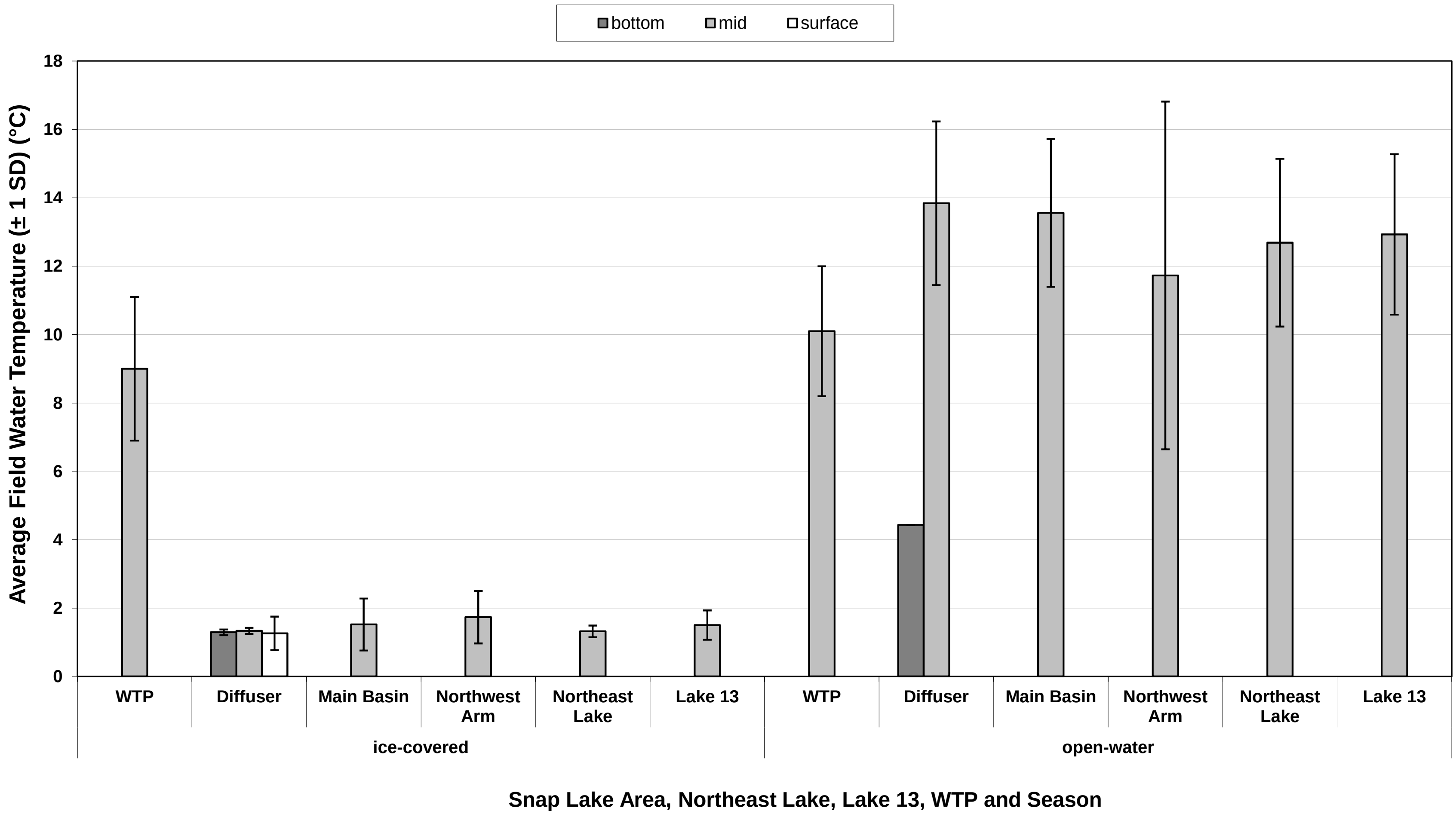


Note: Grey symbols represent Northeast Lake stations; black symbols represent Lake 13 stations; green symbols represent the northwest arm stations; dark blue symbols represent the far-field stations; yellow symbols represent the main basin stations; and, red symbols represent the diffuser stations.

Due to a field error, the September and October water samples and profiles were collected 70 m west of the regular SNP 02-20e monitoring location. Therefore, the temperatures collected on September 4, 2013 to assess seasonal water temperatures (Section 2.4.4), was shown in this figure.

m = metre; °C = degrees Celsius.

Figure 3-43 Average Water Temperature in Snap Lake, Northeast Lake, Lake 13, and Water Treatment Plants, 2013



WTP = water treatment plant (SNP 02-17B); SD = standard deviation; °C = degrees Celsius.

Total Dissolved Solids and Major Ions

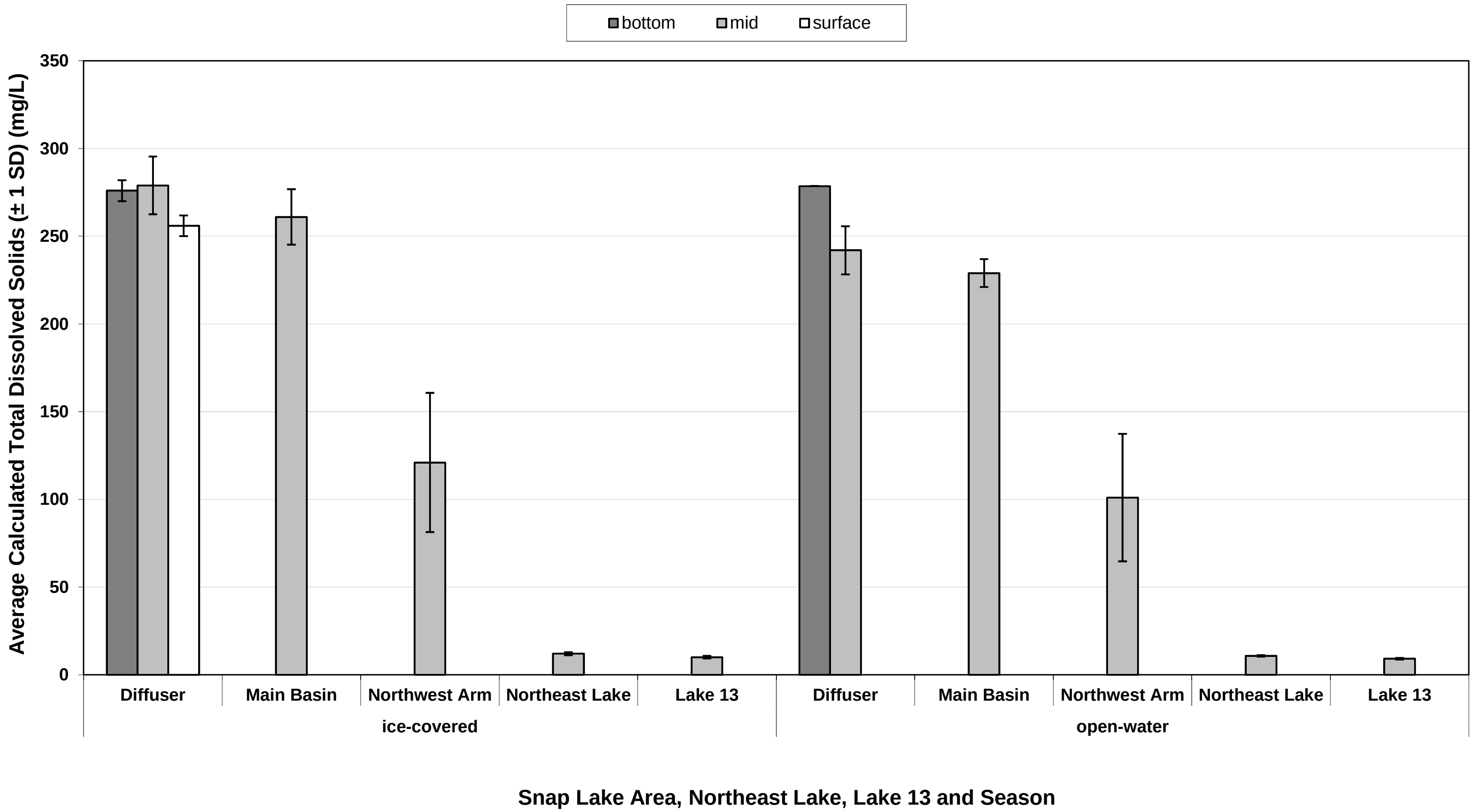
Plots of TDS and three selected major ions (chloride, calcium, and fluoride) show spatial variability between the three main areas of Snap Lake and between Snap Lake and the references lakes in 2013 (Figures 3-44 to 3-47). Average concentrations of TDS and major ions during open-water and ice-covered conditions in the main basin were higher (i.e., by approximately 150 mg/L for TDS) than the northwest arm concentrations, and substantially higher (i.e., by approximately 250 mg/L for TDS) than average concentrations in Northeast Lake and Lake 13. Seasonal variability between open-water and ice-covered conditions was also observed in TDS and major ions in Snap Lake and the reference lakes (Figures 3-44 to 3-47). Average TDS and major ions concentrations were highest during ice-covered conditions; however, the difference between the average TDS and major ions concentrations during ice-covered and open-water conditions was less evident than in 2012 (De Beers 2013b).

Bottom, mid-depth, and surface concentrations of TDS and major ions at the diffuser stations were similar during ice-covered conditions (Figures 3-44 to 3-47). During open-water conditions, the bottom concentration was higher in July at one diffuser location (SNP 02-20e); however, all other locations in the main basin had similar concentrations throughout the water column during open-water conditions. This pattern was consistent with the conductivity profiles, which indicated that the plume may no longer be sinking to the bottom of the lake as it had prior to 2009 (De Beers 2010). The diffuser was designed to discharge the treated effluent away from the bottom of the lake; because the density difference between the plume and lake water has decreased, other factors, such as wind-driven currents, have a greater influence on the plume compared to the effects of relative water densities.

Average TDS and major ions concentrations were higher during ice-covered conditions compared to open-water conditions (Figures 3-44 to 3-47) when mixing was limited to the turbulence caused by the diffuser. During open-water conditions, the lower average concentrations of TDS and major ions were a result of natural processes, such as wind-driven mixing and natural watershed runoff, which contribute to the dilution of major ions concentrations in Snap Lake. However, the difference between ice-covered and open-water concentrations was less evident in 2013, compared to 2012 (De Beers 2013b).

Concentrations of TDS and major ions were generally similar between Northeast Lake and Lake 13 during ice-covered and open-water conditions and were lower than concentrations measured in Snap Lake (Figures 3-44 to 3-47).

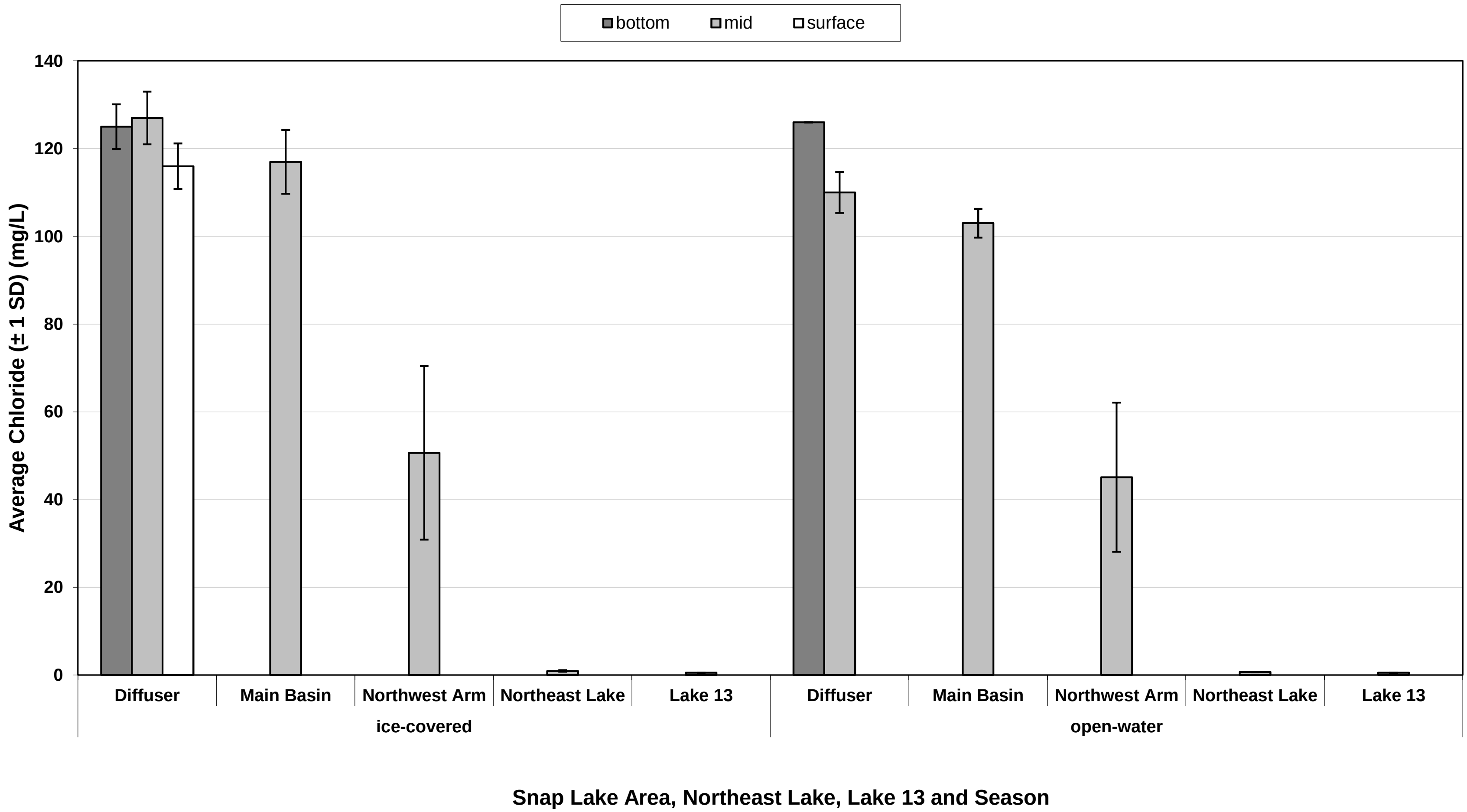
Figure 3-44 Average Calculated Total Dissolved Solids Concentrations (± 1 SD) in Snap Lake, Northeast Lake, and Lake 13, 2013



During the open-water season, the bottom average concentration for the diffuser stations represents the concentration of one sample collected from the bottom of the lake at SNP 02-20e station on July 9, 2013, where the highest conductivity was measured.

SD = standard deviation; mg/L = milligrams per litre.

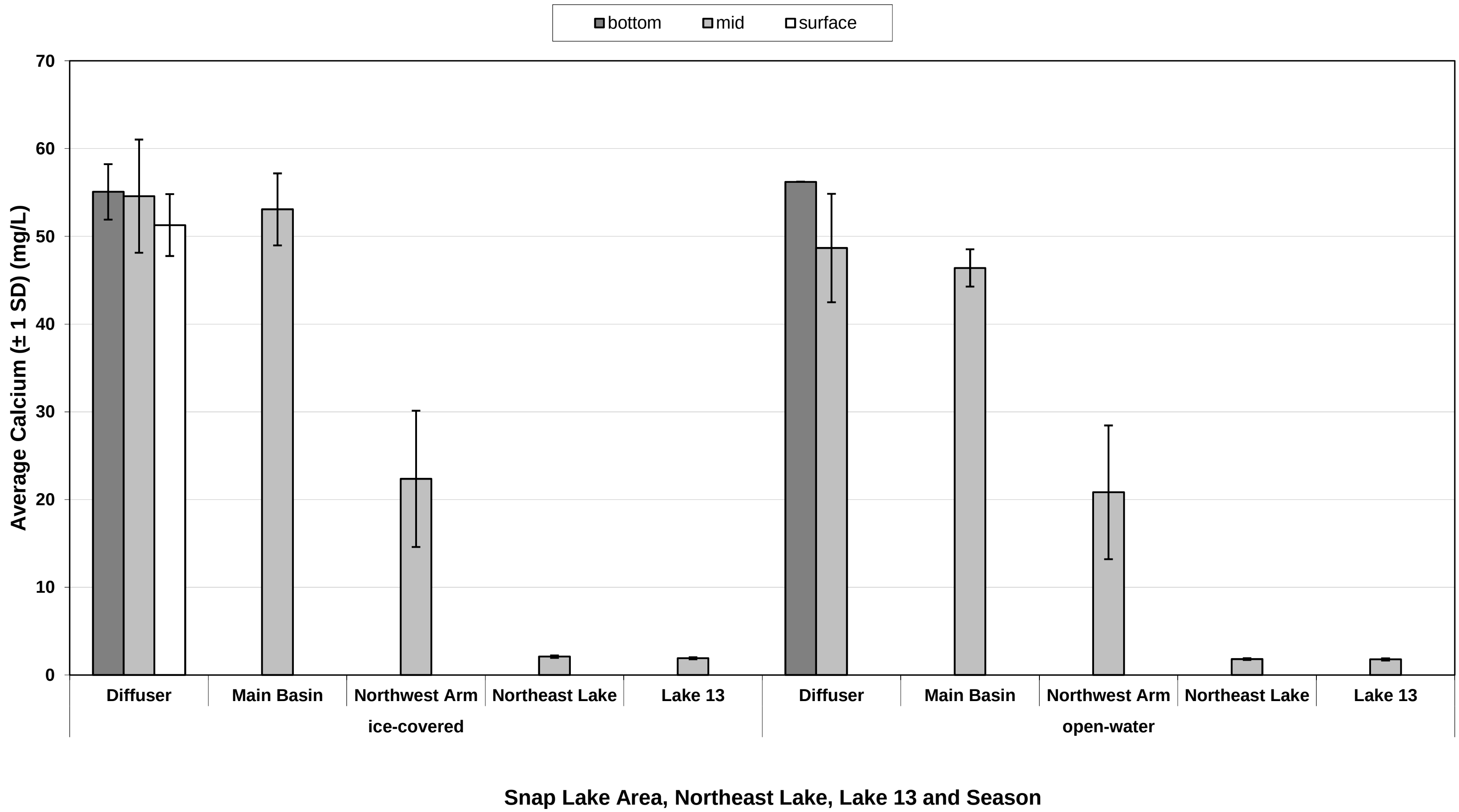
Figure 3-45 Average Chloride Concentrations (± 1 SD) in Snap Lake, Northeast Lake, and Lake 13, 2013



During the open-water season, the bottom average concentration for the diffuser stations represents the concentration of one sample collected from the bottom of the lake at SNP 02-20e station on July 9, 2013, where the highest conductivity was measured.

SD = standard deviation; mg/L = milligrams per litre.

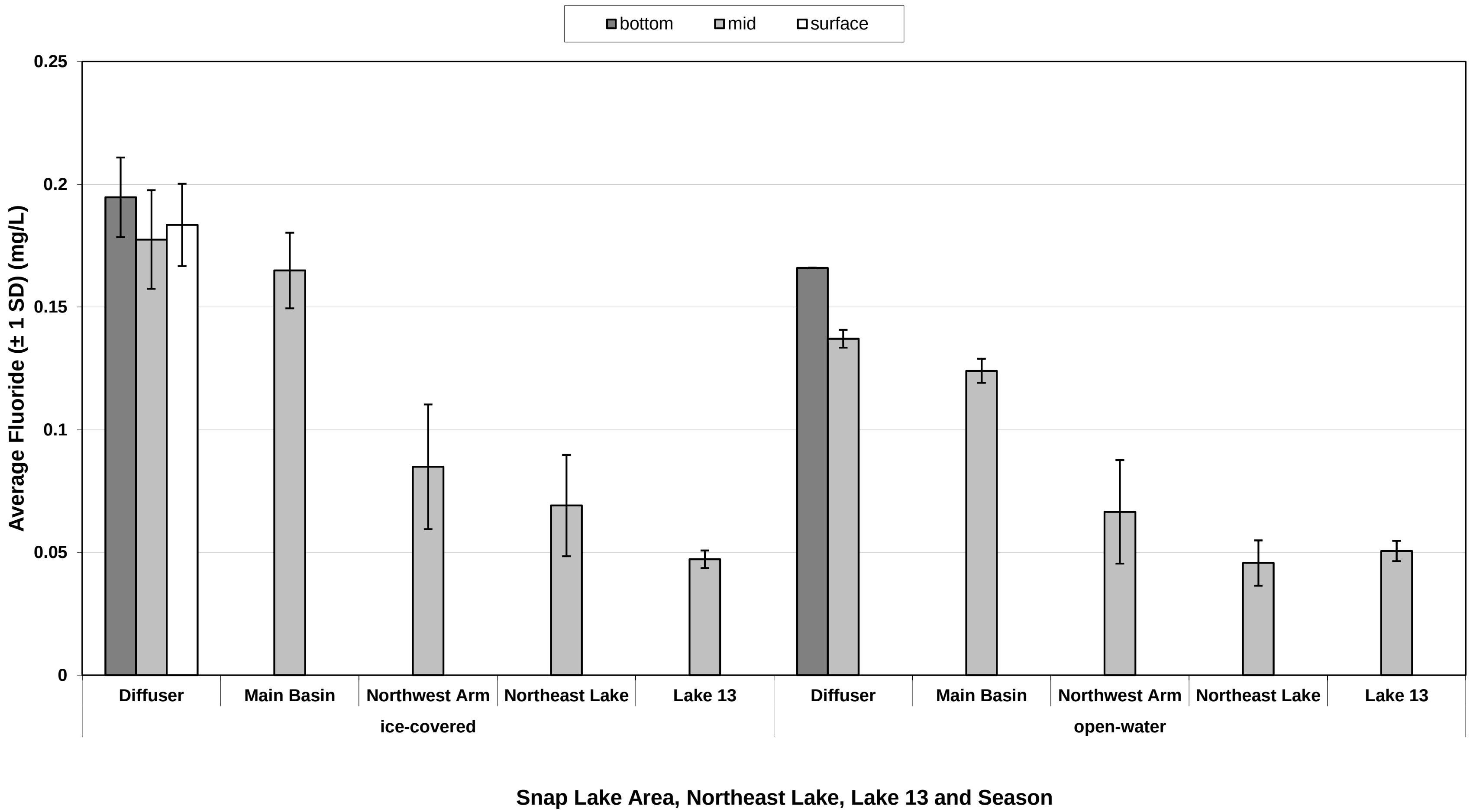
Figure 3-46 Average Calcium Concentrations (± 1 SD) in Snap Lake, Northeast Lake, and Lake 13, 2013



During the open-water season, the bottom average concentration for the diffuser stations represents the concentration of one sample collected from the bottom of the lake at SNP 02-20e station on July 9, 2013, where the highest conductivity was measured.

SD = standard deviation; mg/L = milligrams per litre.

Figure 3-47 Average Fluoride Concentrations (± 1 SD) in Snap Lake, Northeast Lake, and Lake 13, 2013



During the open-water season, the bottom average concentration for the diffuser stations represents the concentration of one sample collected from the bottom of the lake at SNP 02-20e station on July 9, 2013, where the highest conductivity was measured.

SD = standard deviation; mg/L = milligrams per litre.

Nutrients

Spatial variability was observed in the nitrogen parameters (nitrate, nitrite, ammonia, TN, and TKN) in Snap Lake. Concentrations of nitrate, nitrite, TKN, TN, and ammonia in Snap Lake decreased with distance from the diffuser (i.e., highest at the diffuser stations and lowest at the northwest arm stations; Figures 3-48 to 3-51, and Figure 3H-20). Higher concentrations of nutrients were expected in areas closer to the diffuser, because minewater from underground and domestic waste water contain elevated concentrations of both nitrogen and phosphorus (Appendix 3G).

Greater variability was observed in the TN results in the euphotic zone at the diffuser stations and in the northwest arm stations compared to the euphotic zone in the main basin and in reference lakes. However, differences between average TN concentrations collected at mid-depth and within the euphotic zone (i.e., depth-integrated samples collected at 1 m in the upper 6 m of the water column) were not observed in either Snap Lake or in the reference lakes in 2013.

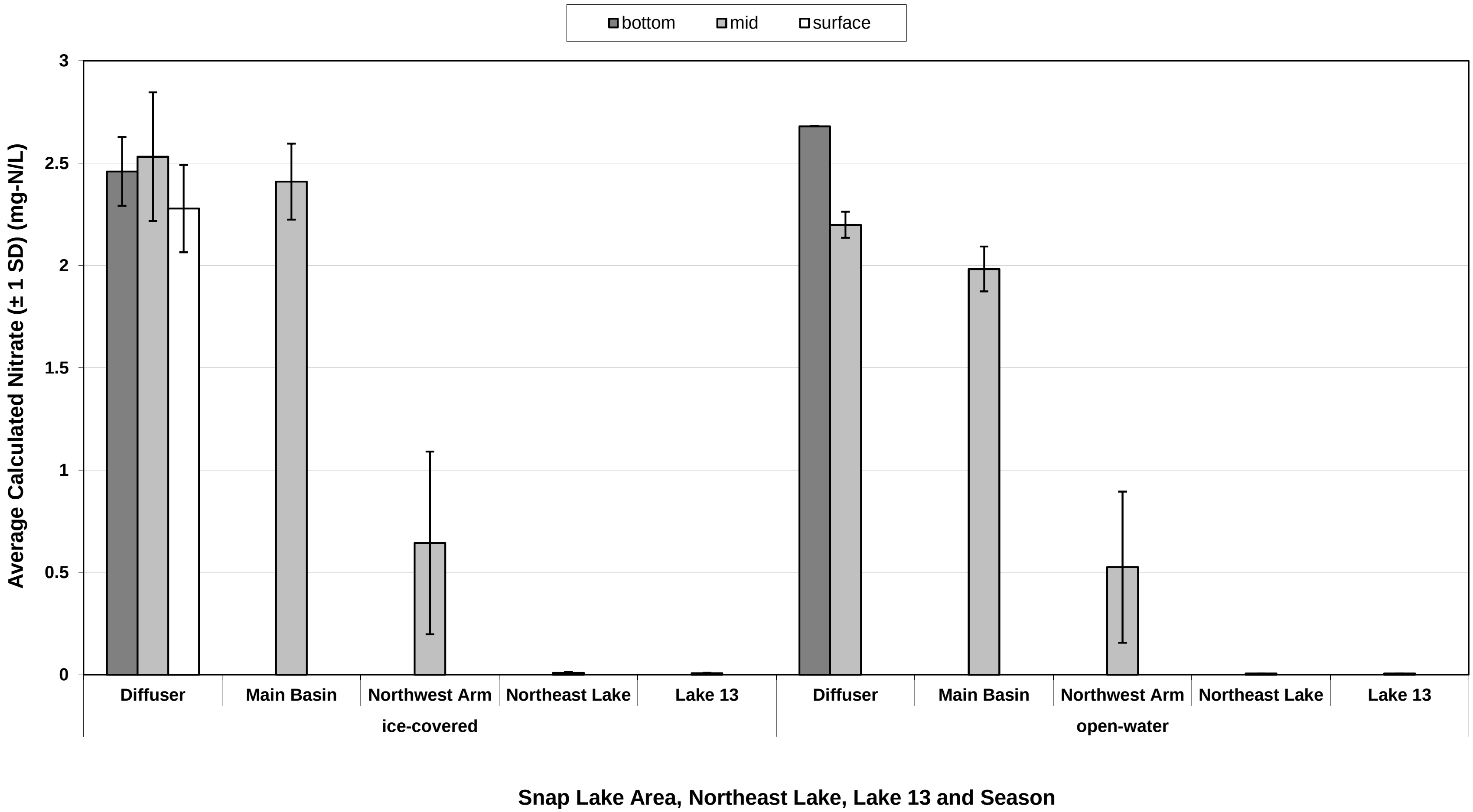
Seasonal differences occurred in TKN, TN, ammonia, and nitrate concentrations in Snap Lake in 2013; average concentrations of these parameters were higher during ice-covered conditions compared to open-water conditions (Figures 3-49 and 3-51). Nitrification of ammonia to nitrate by bacteria and assimilation of ammonia and nitrate by phytoplankton could contribute to the decrease in ammonia and nitrate concentrations during open-water conditions. Limited assimilation and slower nitrification rates are expected during ice-covered conditions resulting from factors including colder temperatures, lower DO concentrations, and shorter periods of light to encourage phytoplankton productivity. Similar to the seasonal patterns observed for TDS and major ions, the 2013 seasonal differences in nitrogen parameters are less evident than in previous years (De Beers 2010, 2011, 2012b, 2013b).

Clear spatial patterns in TP were not evident during ice-covered or open-water conditions (Figure 3-52), consistent with previous years (De Beers 2010, 2011, 2012b, 2013b). Differences between average TP concentrations at mid-depth and within the euphotic zones were not discernable in 2013. The lack of a distinct spatial pattern in phosphorus may be related to the uncertainty associated with: low-level phosphorus concentrations in Snap Lake (Appendix 3B); rapid uptake of phosphorus by phytoplankton which reside at shallower depths within the euphotic zone; sedimentation processes; or, a combination of all three possible causes (Figures 3-52).

Greater variability in TP concentrations were observed in Snap Lake and the reference lakes during open-water conditions, which may be related to an increase in biological processes that influence phosphorus concentrations in the water column.

Concentrations of nutrients were typically similar between Northeast Lake and Lake 13 with the exception of TP; TP concentrations were higher in Lake 13 during open-water conditions (Figures 3-48 to 3-52). Two elevated TP concentrations (i.e., 0.021 and 0.054 mg/L) caused the higher average and larger standard error shown for the open-water euphotic zone TP in Lake 13 (Figure 3-52); all other 2013 Lake 13 values for euphotic zone samples were 0.006 mg/L or less. Data quality issues were not identified in the elevated TP samples from Lake 13, so results were either outliers or an indication of greater variability in TP in Lake 13 relative to Snap Lake and Northeast Lake; future monitoring of TP in Lake 13 will help to identify which explanation is more probable.

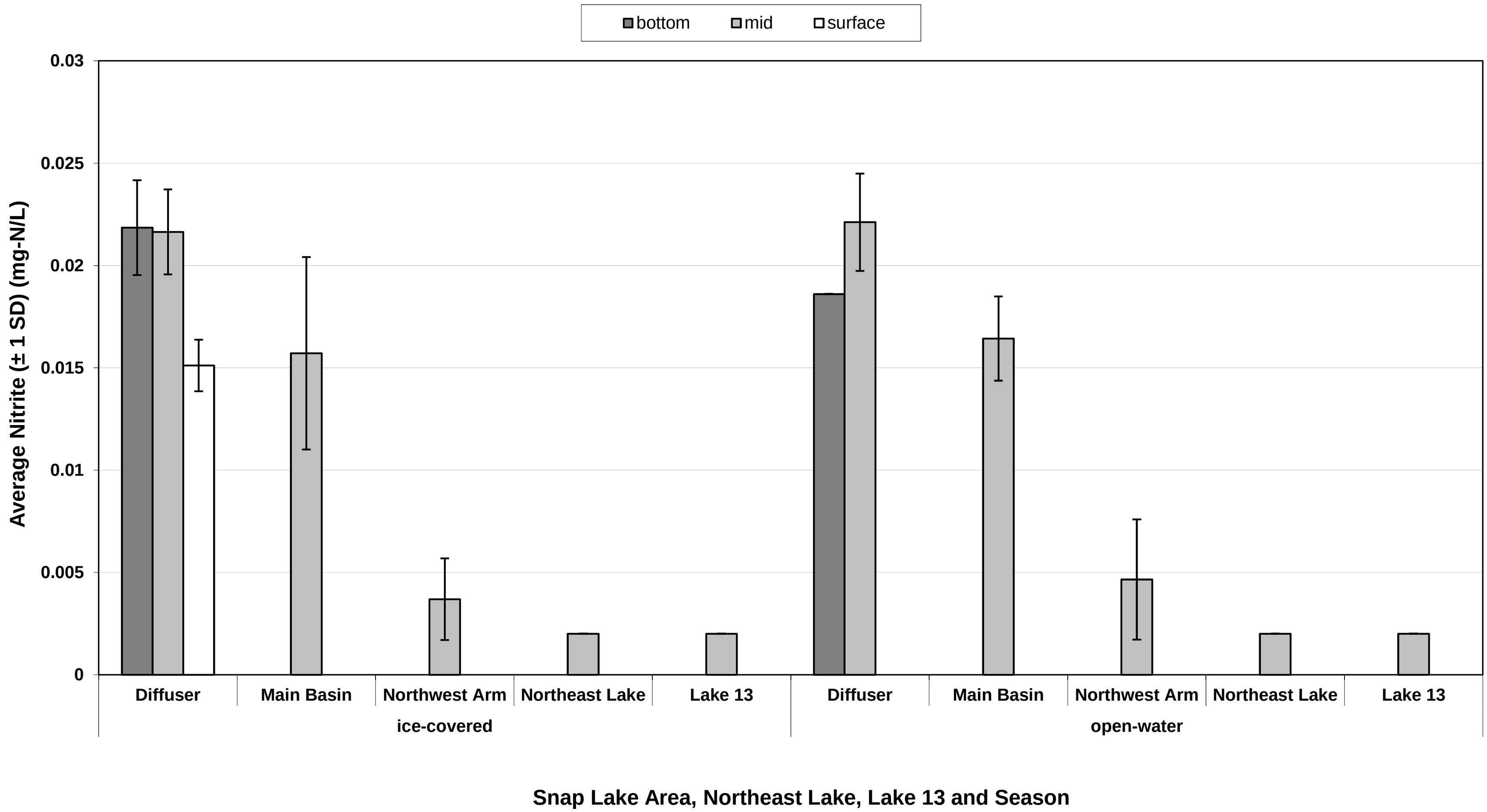
Figure 3-48 Average Calculated Nitrate Concentrations (± 1 SD) in Snap Lake, Northeast Lake, and Lake 13, 2013



During the open-water season, the bottom average concentration for the diffuser stations represents the concentration of one sample collected from the bottom of the lake at SNP 02-20e station on July 9, 2013, where the highest conductivity was measured.

SD = standard deviation; mg-N/L = milligrams as nitrogen per litre

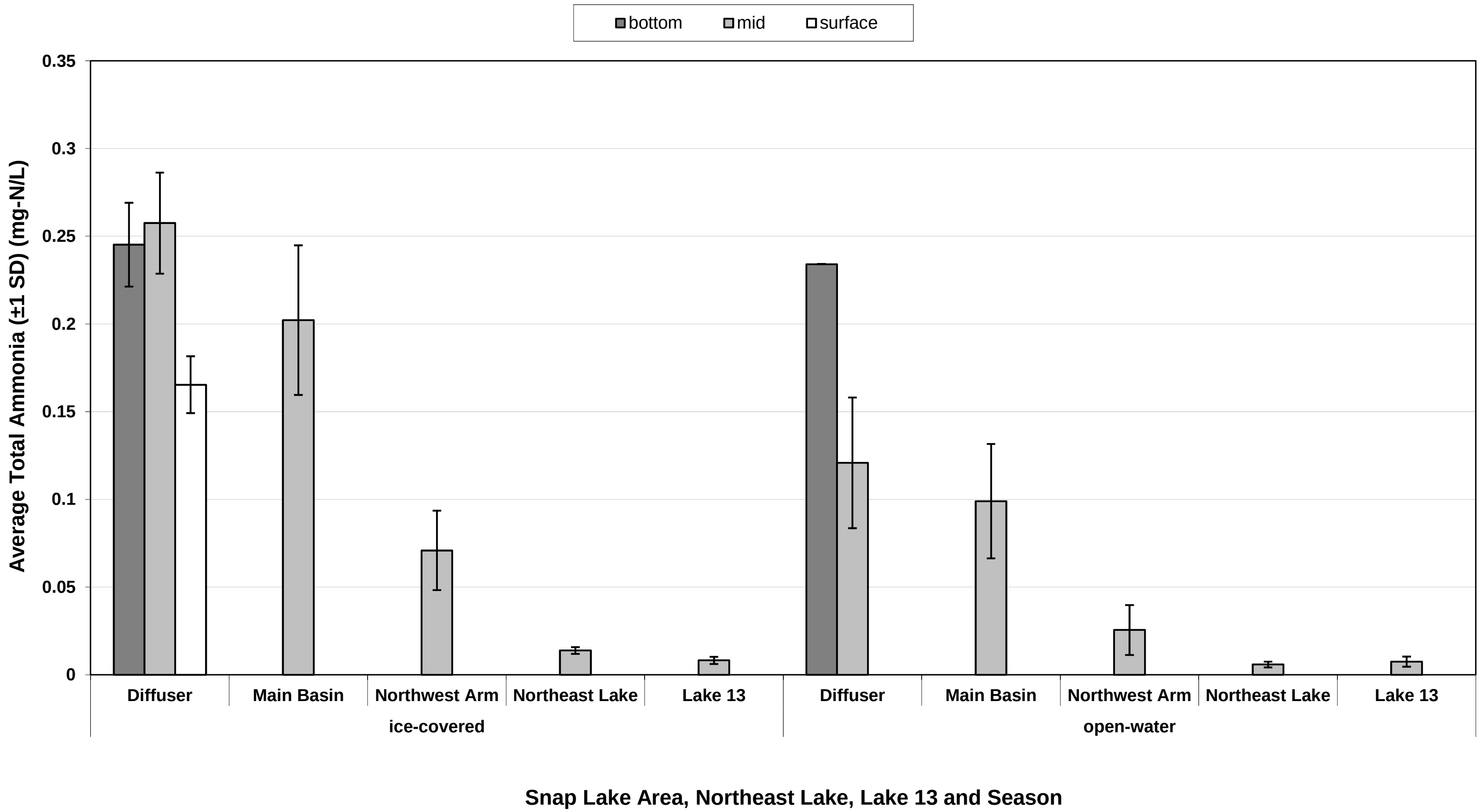
Figure 3-49 Average Nitrite Concentrations (± 1 SD) in Snap Lake, Northeast Lake, and Lake 13, 2013



During the open-water season, the bottom average concentration for the diffuser stations represents the concentration of one sample collected from the bottom of the lake at SNP 02-20e station on July 9, 2013, where the highest conductivity was measured.

SD = standard deviation; N = nitrogen; mg/L = milligrams per litre.

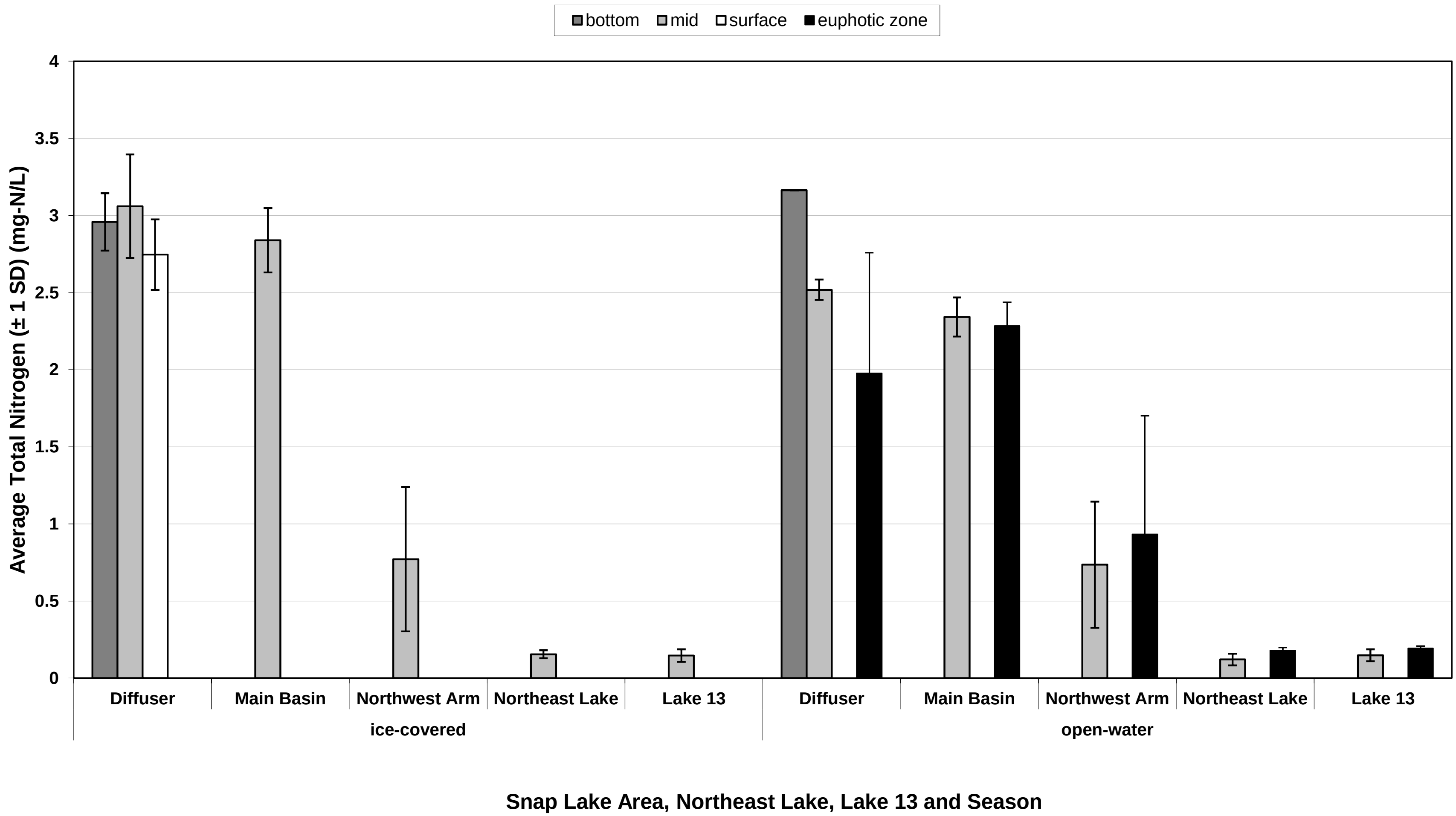
Figure 3-50 Average Total Ammonia Concentrations (± 1 SD) in Snap Lake, Northeast Lake, and Lake 13, 2013



During the open-water season, the bottom average concentration for the diffuser stations represents the concentration of one sample collected from the bottom of the lake at SNP 02-20e station on July 9, 2013, where the highest conductivity was measured.

SD = standard deviation; N = nitrogen; mg/L = milligrams per litre.

Figure 3-51 Average Total Nitrogen Concentrations (± 1 SD) in Snap Lake, Northeast Lake, and Lake 13, 2013

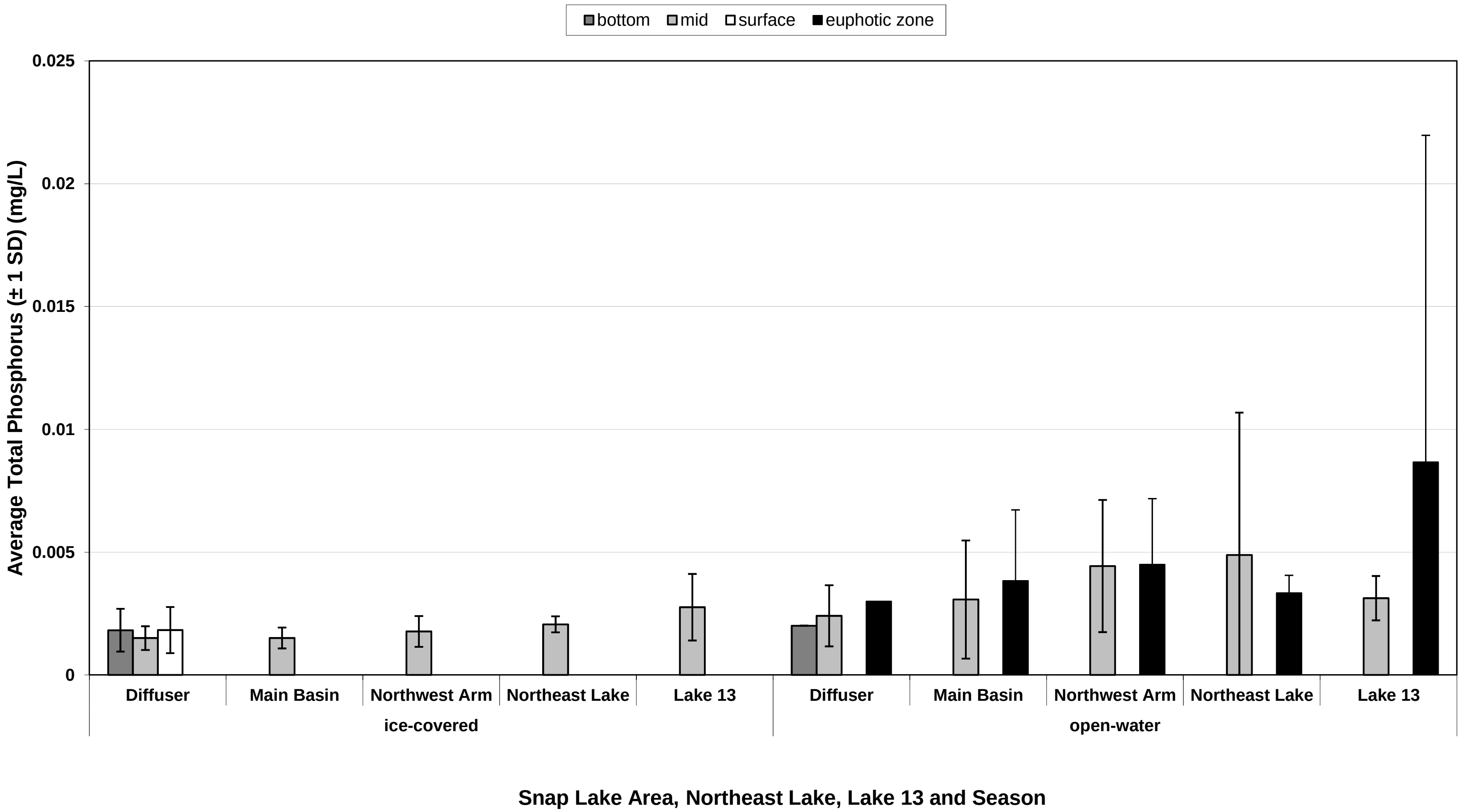


During the open-water season, the bottom average concentration for the diffuser stations represents the concentration of one sample collected from the bottom of the lake at SNP 02-20e station on July 9, 2013, where the highest conductivity was measured.

Euphotic zone (depth-integrated) samples were collected as part of the plankton, and analyzed by UofA.

SD = standard deviation; mg-N/L = milligrams as nitrogen per litre.

Figure 3-52 Average Total Phosphorus Concentrations (± 1 SD) in Snap Lake, Northeast Lake, and Lake 13, 2013



During the open-water season, the bottom average concentration for the diffuser stations represents the concentration of one sample collected from the bottom of the lake at SNP 02-20e station on July 9, 2013, where the highest conductivity was measured.

Euphotic zone (depth-integrated) samples were collected as part of the plankton component and analyzed by U of A.

SD = standard deviation; mg-P/L = milligrams as phosphorus per litre.

Metals

Spatial and seasonal variability were evident for some of the metals. However, variability was less prominent in 2013, and no clear spatial patterns were observed for most metals compared to previous years (De Beers 2010, 2011, 2012b, 2013b). Plots for representative metals are presented in Figures 3-53 to 3-62; plots for the other metals are presented in Appendix 3H. Metals with strong positive correlations to conductivity (e.g., boron, barium, lithium, molybdenum, nickel, rubidium, strontium, uranium; see Section 3.4.4), continued to demonstrate clear differences in concentrations between the main basin and northwest arm of Snap Lake relative to those metals with weak correlations to conductivity (e.g., copper, cobalt, iron, lead, mercury, titanium; Table 3-16). Spatial and seasonal trends for total metals are summarized in Table 3-16.

Table 3-16 Summary of Spatial and Seasonal Trends for Total Metals Measured in Snap Lake, 2013

Spatial/Seasonal Pattern	Total Metals that Apply	Example Plots[a]
Spatial Pattern		
Average concentrations were clearly higher in the main basin compared to the northwest arm	Boron, barium, lithium, molybdenum, nickel, rubidium, strontium, and uranium[b]	Total barium, Figure 3-53 Total molybdenum, Figure 3-54 Total strontium, Figure 3-55
Average concentrations were elevated at diffuser stations, but much lower in the main basin	Cadmium (ice-covered), chromium, mercury (ice-covered)[c], and titanium[c]	Total chromium, Figure 3-56
No clear spatial pattern in Snap Lake, but average concentrations in Snap Lake were higher than average reference lake concentrations	Cobalt[c], and manganese	Total cobalt, Figure 3-57 Total manganese, Figure 3-58
Average concentrations in the reference lakes were higher than Snap Lake	Arsenic (in Lake 13)	Total arsenic, Figure 3-59
No clear spatial pattern	Aluminum, antimony[d], copper[c], iron[c], lead[c], and zinc Beryllium, bismuth, cesium, hexavalent chromium, methyl mercury, selenium, silver, thallium, and vanadium were all at or near the detection limit in 2013.	Total antimony, Figure 3-60 Total iron, Figure 3-61

Table 3-16 Summary of Spatial and Seasonal Trends for Total Metals Measured in Snap Lake, 2013

Spatial/Seasonal Pattern	Total Metals that Apply	Example Plots[a]
Seasonal Pattern		
Average lake concentrations were higher during ice-covered conditions	Antimony [d]	Total antimony, Figure 3-60
Average lake concentrations were higher during open-water conditions	Aluminum, iron[c], and zinc	Total iron, Figure 3-61
No clear seasonal pattern	Arsenic, barium[b], boron[b], cadmium, chromium, cobalt[c], copper[c],lead[c], lithium[b], manganese, mercury[c], molybdenum[b] , nickel[b], rubidium[b], strontium[b], titanium[c], and uranium[b] Beryllium, bismuth, cesium, hexavalent chromium, methyl mercury, selenium, silver, thallium, and vanadium were all at or near the detection limit in 2013.	Total arsenic, Figure 3-59 Total molybdenum, Figure 3-54 Total copper, Figure 3-62

a) Examples of total metals representing the spatial and seasonal trends, or lack of trend, are shown in Figures 3-53 to 3-62. The remaining metals are presented in Appendix 3H.

b) Metals in the list were strongly positive correlated with conductivity (Section 3.4.4).

c) Metals in the list were weakly correlated with conductivity (Section 3.4.3).

d) High surface values were measured, but there may have been contamination issues that contributed to these values.

Cobalt and manganese appeared to be higher in Snap Lake relative to the reference lakes in 2013, even though they were not identified as strongly correlated with conductivity, which was used an indicator of treated effluent. However, cobalt and manganese are present in the effluent at concentrations similar to other parameters that are correlated with conductivity (Appendix 3D). Manganese and cobalt may be behaving less conservatively than other effluent-related parameters within Snap Lake because of oxidation (in oxygenated waters) and reduction processes (in anoxic waters) that can cause manganese, and associated metals such as cobalt, to move between insoluble forms (e.g., manganese oxides) that can settle out of the water column and soluble forms that result in manganese being released to the water column, respectively. For example, elevated manganese in the northwest arm was likely related to lower DO in the northwest arm (Section 3.4.3; Figure 3-7, panels a and b), because manganese concentrations were highest at the deepest station in the northwest arm (SNAP20B) during ice-covered and open-water conditions where DO concentrations were lowest (Appendix 3D).

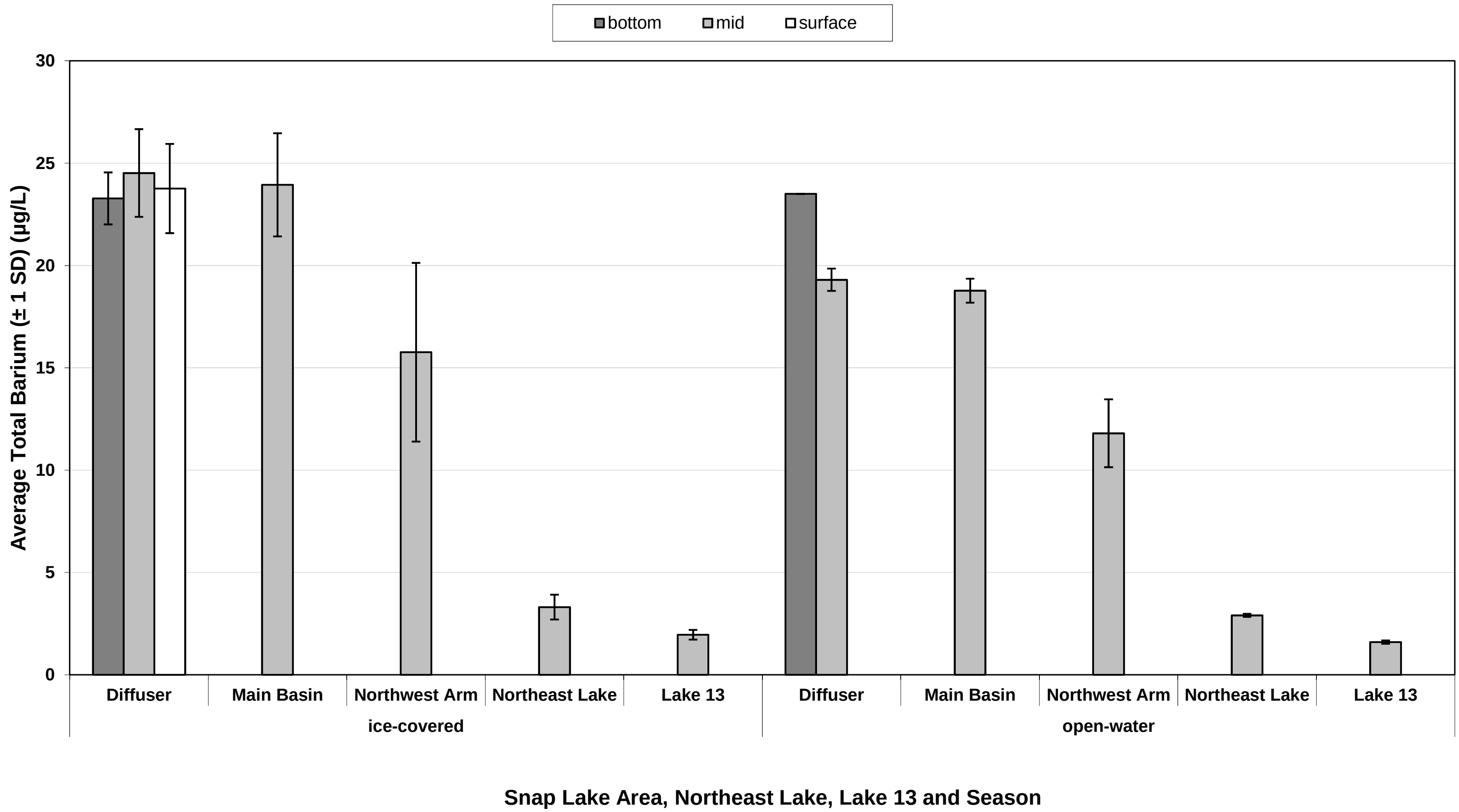
Antimony concentrations were higher during ice-covered conditions (Figure 3-60); elevated concentrations were mainly at the surface of the water column. The anomalous pattern in antimony may be due to sample contamination, as discussed in Appendix 3A and Section 3.4.7.

Aluminum, iron, and zinc concentrations were higher during open-water conditions. Higher concentrations of iron and zinc may be related to the inflows from Stream S27 during open-water conditions. When the highest concentrations of total iron and zinc were measured in Stream S27 in September (i.e., 806 µg/L and 4.6 µg/L, respectively), high concentrations in the Snap Lake for total iron and zinc were also measured at SNAP02A (i.e., 12 µg/L) and at SNAP29 (i.e., 2.2 µg/L), respectively in the northwest arm of

Snap Lake (Section 3.4.6). Iron and aluminum are often associated with suspended particles; the observed increases in aluminum and iron during open-water conditions may be related to natural increases in TSS loadings to Snap Lake during open-water conditions from freshet and subsequent runoff events. For example, elevated aluminum concentrations were observed in Streams S27 and S1 (range of 43 to 138 µg/L) relative to Snap Lake (range of 1 to 8 µg/L) during open-water conditions.

Concentrations of metals were typically similar or lower in the reference lakes when compared with Snap Lake, with the exceptions of arsenic at Lake 13 during both open-water and ice-covered conditions. Concentrations of metals were typically similar between Northeast Lake and Lake 13, with the exception of total arsenic (Figure 3-59).

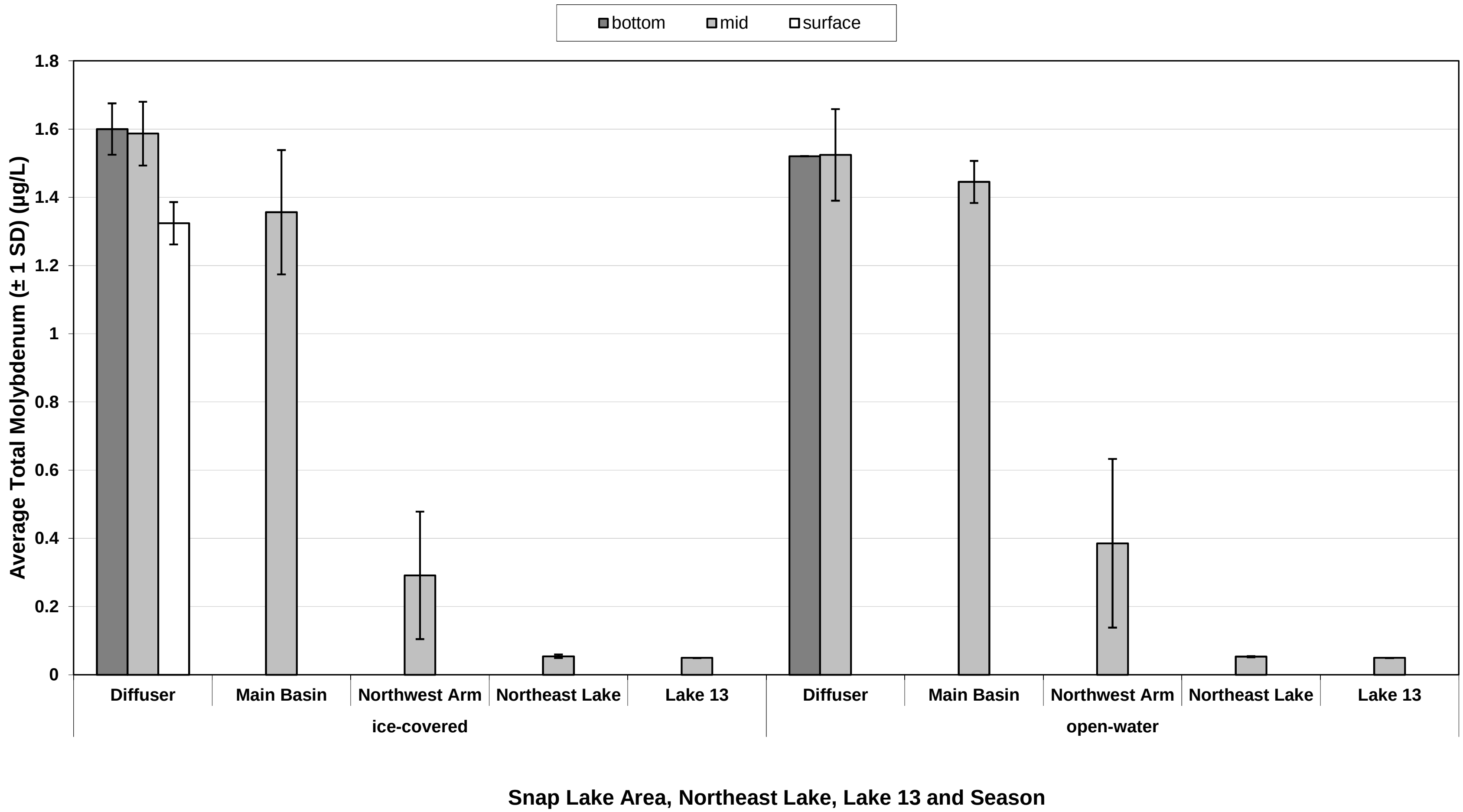
Figure 3-53 Average Total Barium Concentrations in Snap Lake, Northeast Lake, and Lake 13, 2013



During the open-water season, the bottom average concentration for the diffuser stations represents the concentration of one sample collected from the bottom of the lake at SNP 02-20e station on July 9, 2013, where the highest conductivity was measured.

SD = standard deviation; µg/L = micrograms per litre.

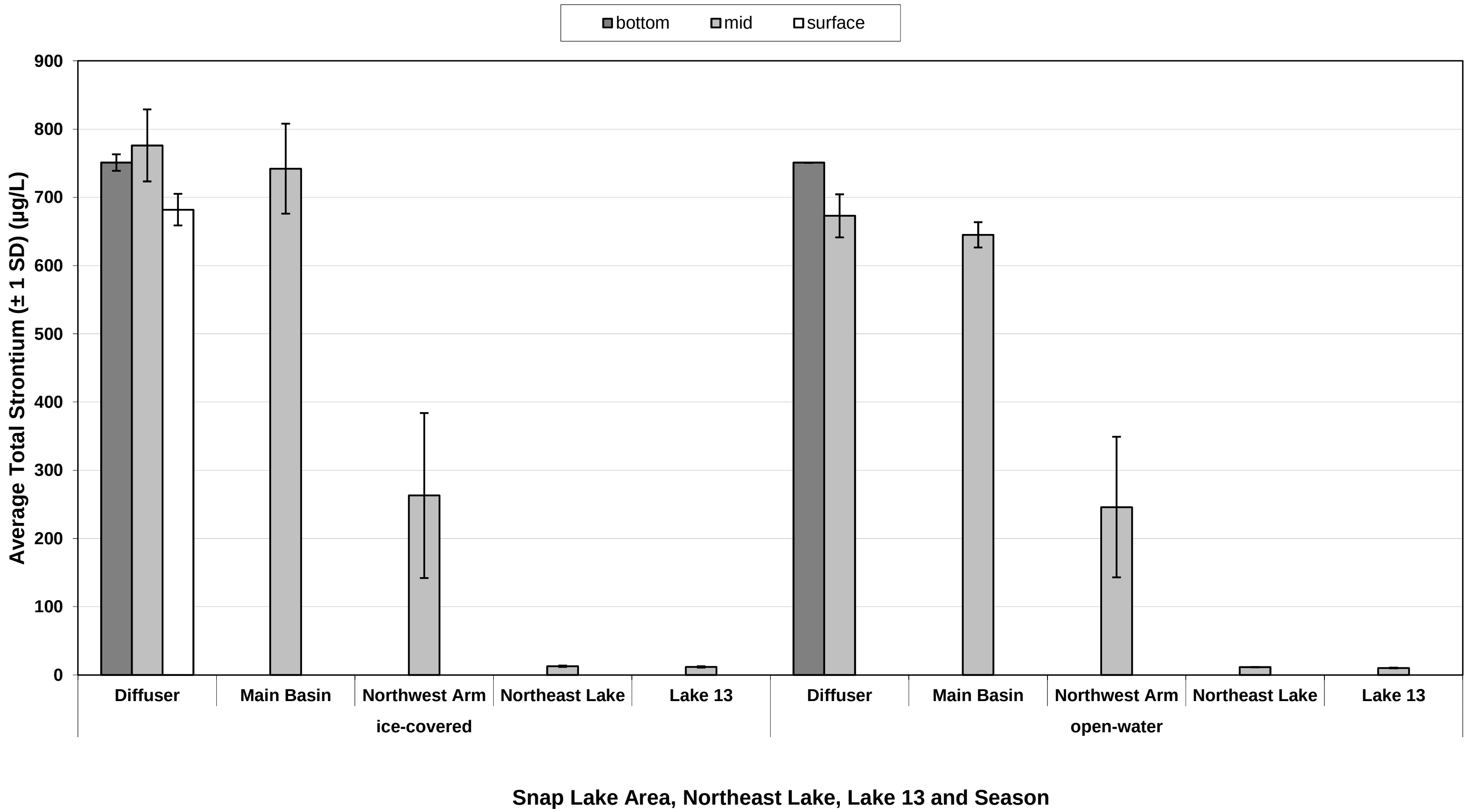
Figure 3-54 Average Total Molybdenum Concentrations (± 1 SD) in Snap Lake, Northeast Lake, and Lake 13, 2013



During the open-water season, the bottom average concentration for the diffuser stations represents the concentration of one sample collected from the bottom of the lake at SNP 02-20e station on July 9, 2013, where the highest conductivity was measured.

SD = standard deviation; µg/L = micrograms per litre.

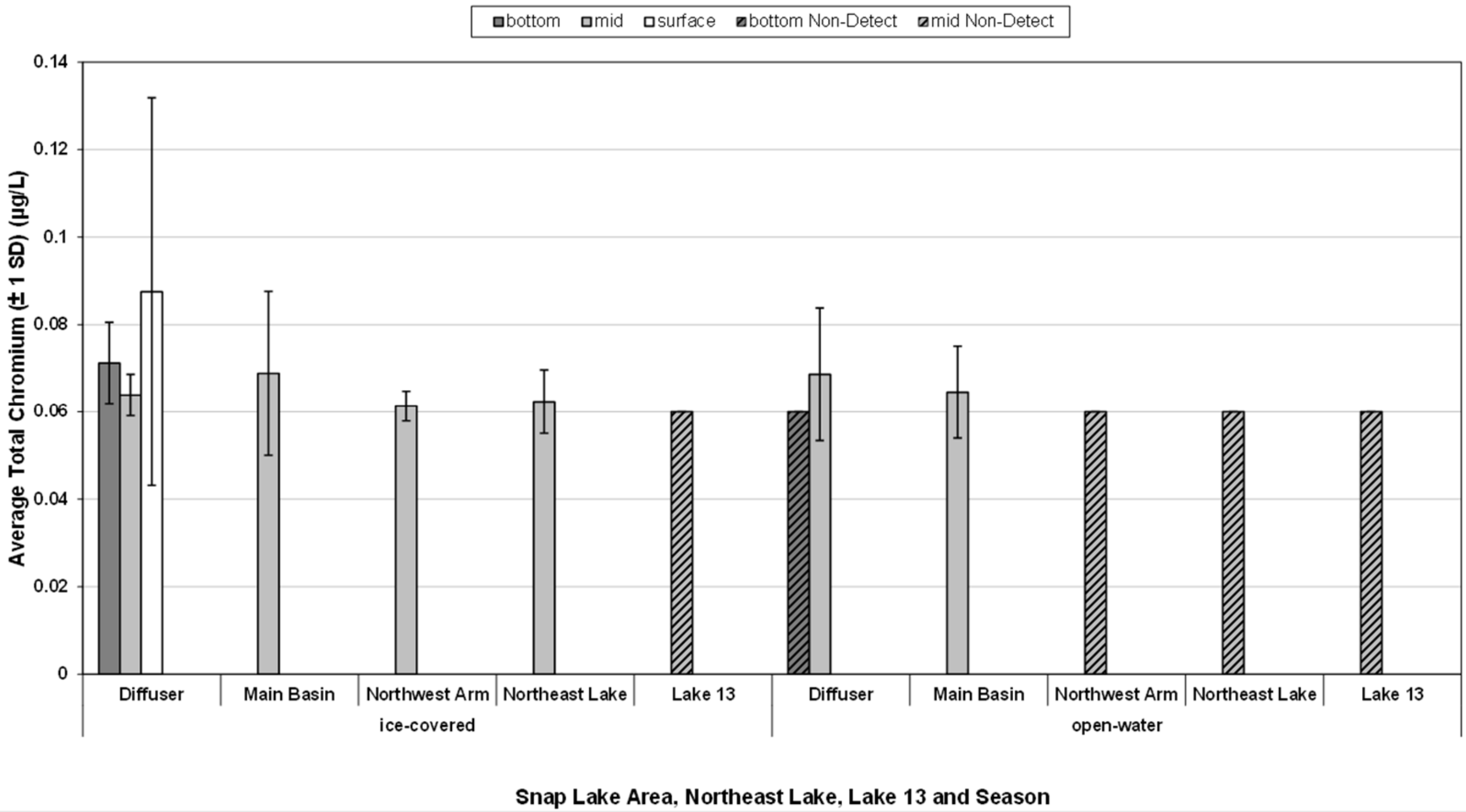
Figure 3-55 Average Total Strontium Concentrations (± 1 SD) in Snap Lake, Northeast Lake, and Lake 13, 2013



During the open-water season, the bottom average concentration for the diffuser stations represents the concentration of one sample collected from the bottom of the lake at SNP 02-20e station on July 9, 2013, where the highest conductivity was measured.

SD = standard deviation; µg/L = micrograms per litre.

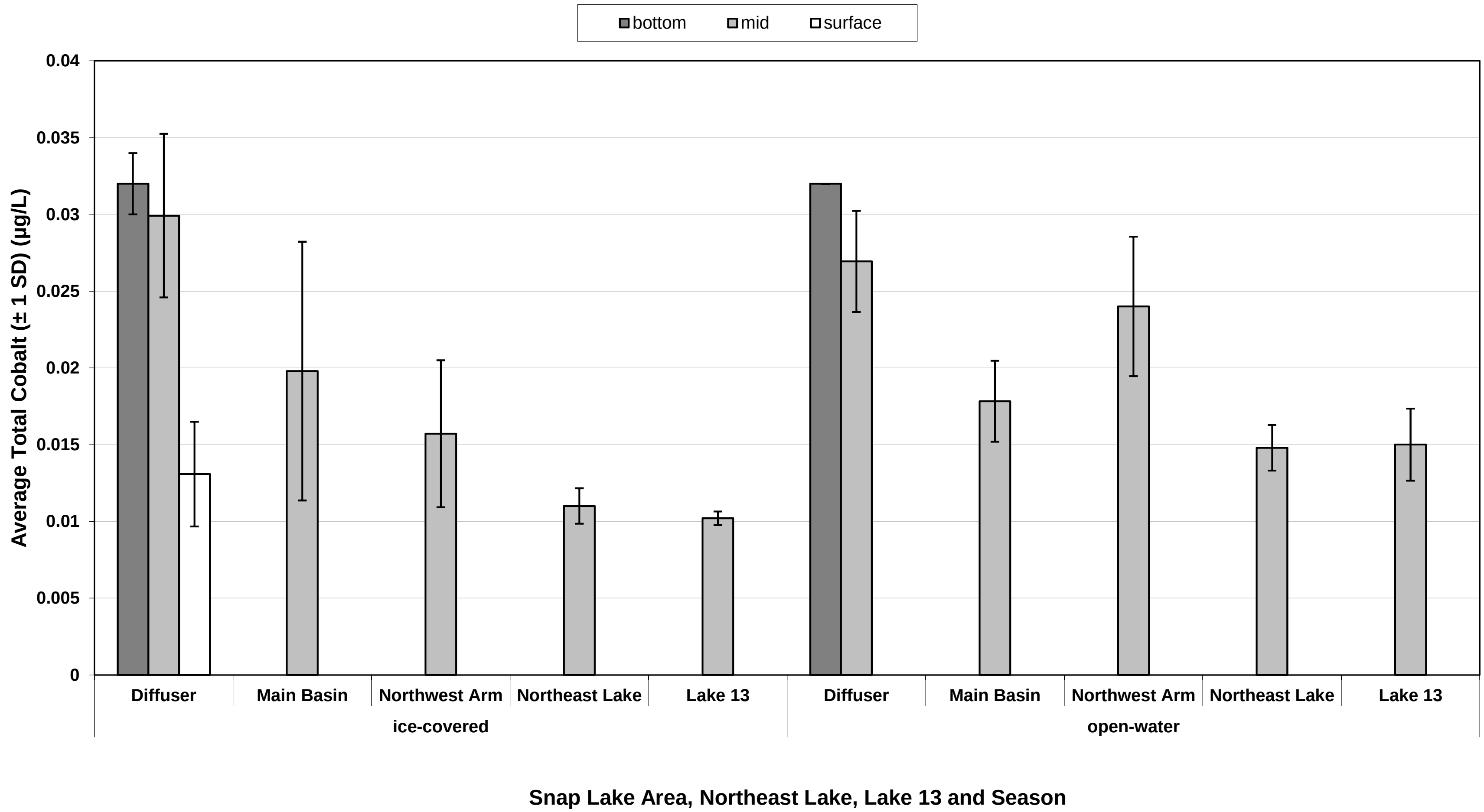
Figure 3-56 Average Total Chromium Concentrations (± 1 SD) in Snap Lake, Northeast Lake, and Lake 13, 2013



During the open-water season, the bottom average concentration for the diffuser stations represents the concentration of one sample collected from the bottom of the lake at SNP 02-20e station on July 9, 2013, where the highest conductivity was measured.

SD = standard deviation; µg/L = micrograms per litre.

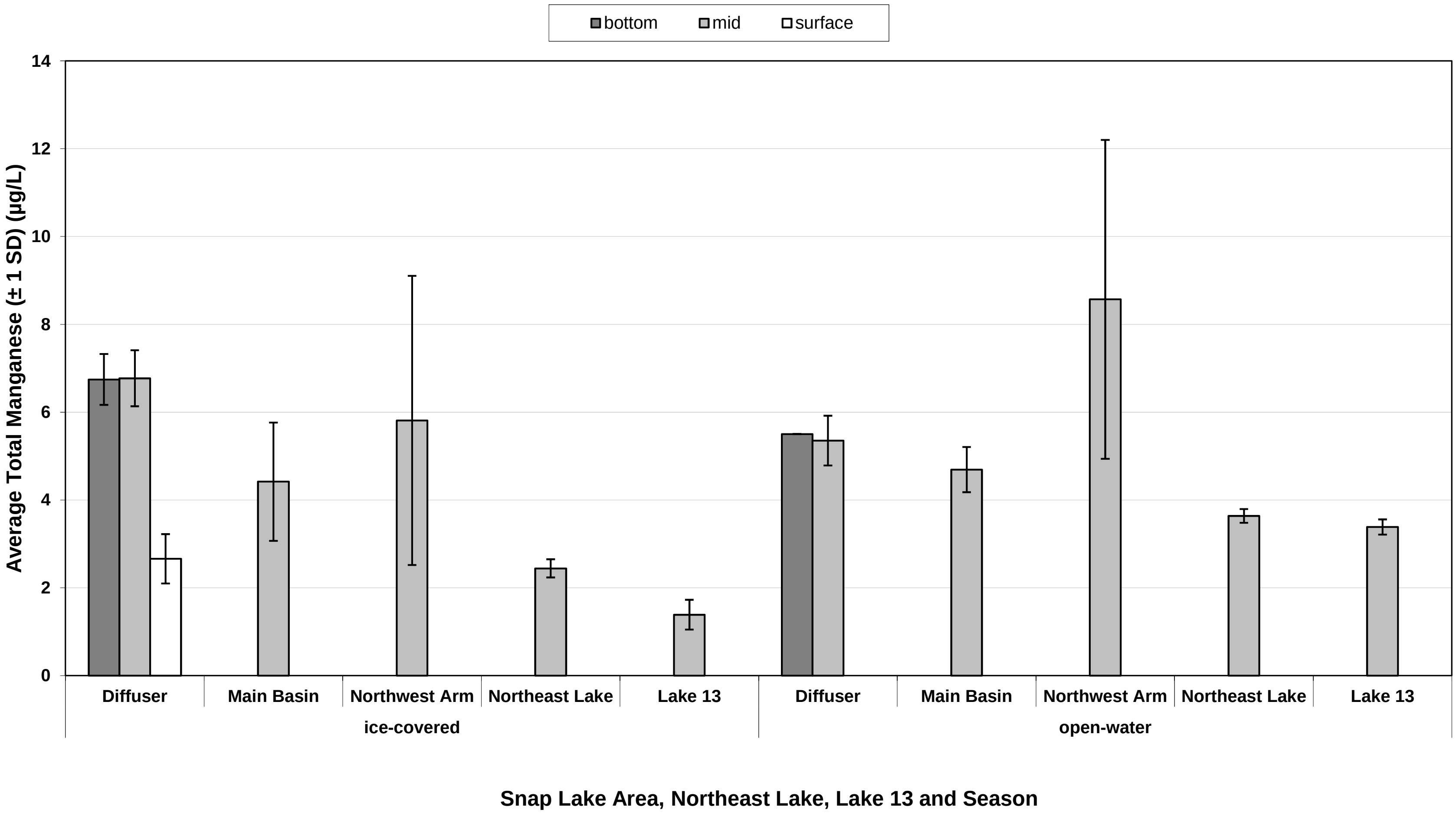
Figure 3-57 Average Total Cobalt Concentrations (± 1 SD) in Snap Lake, Northeast Lake, and Lake 13, 2013



During the open-water season, the bottom average concentration for the diffuser stations represents the concentration of one sample collected from the bottom of the lake at SNP 02-20e station on July 9, 2013, where the highest conductivity was measured.

SD = standard deviation; µg/L = micrograms per litre.

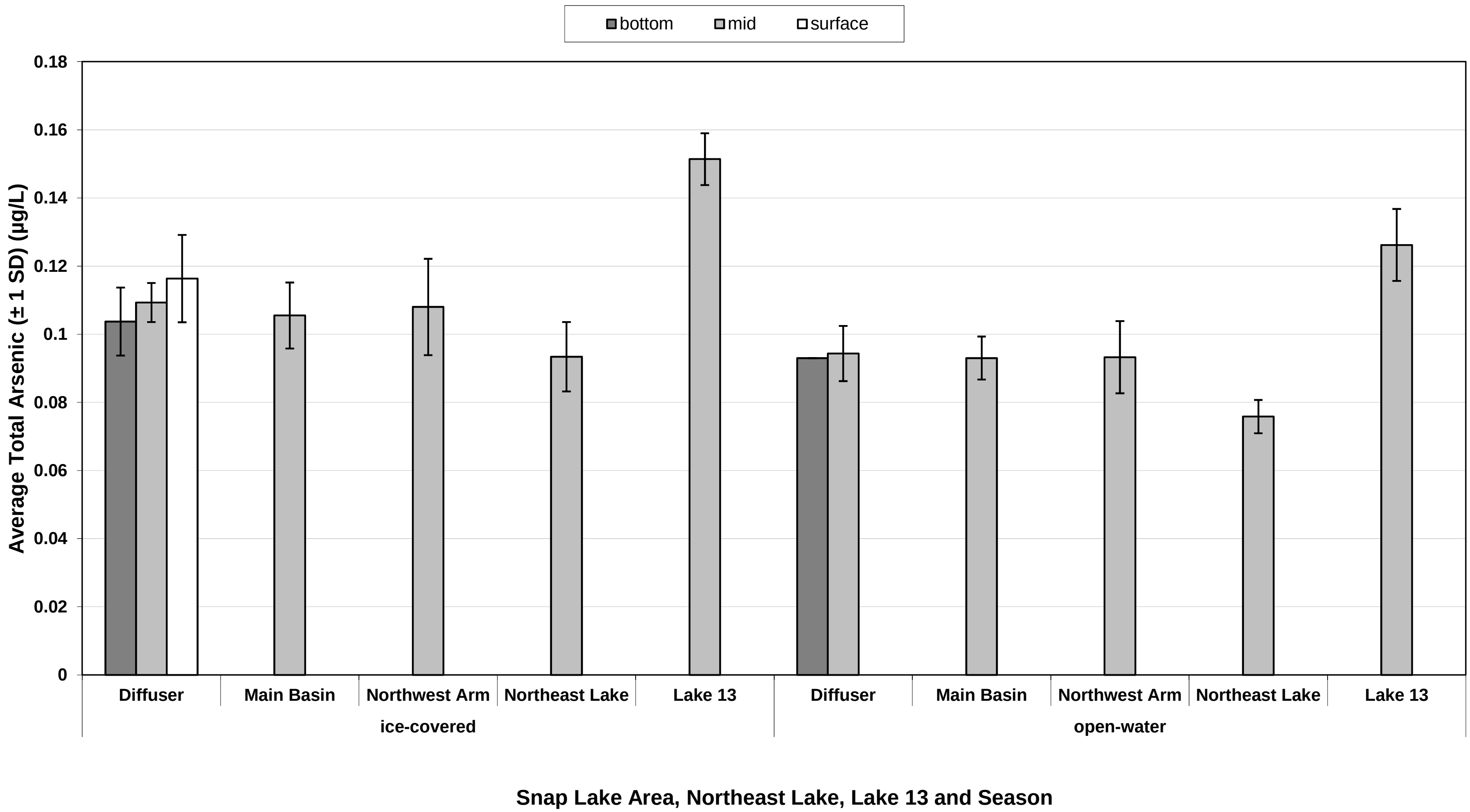
Figure 3-58 Average Total Manganese Concentrations (± 1 SD) in Snap Lake, Northeast Lake, and Lake 13, 2013



During the open-water season, the bottom average concentration for the diffuser stations represents the concentration of one sample collected from the bottom of the lake at SNP 02-20e station on July 9, 2013, where the highest conductivity was measured.

SD = standard deviation; µg/L = micrograms per litre.

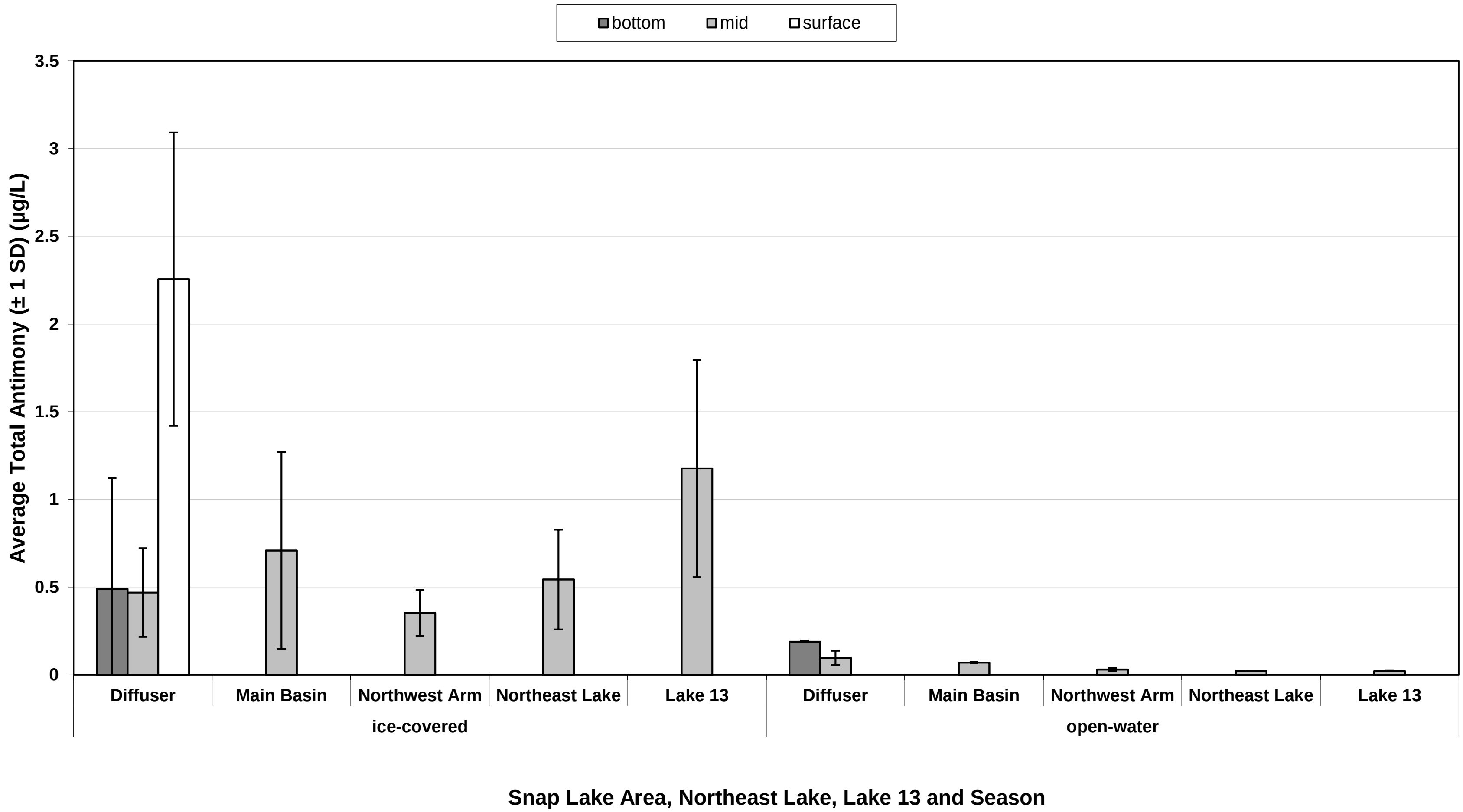
Figure 3-59 Average Total Arsenic Concentrations (± 1 SD) in Snap Lake, Northeast Lake, and Lake 13, 2013



During the open-water season, the bottom average concentration for the diffuser stations represents the concentration of one sample collected from the bottom of the lake at SNP 02-20e station on July 9, 2013, where the highest conductivity was measured.

SD = standard deviation; µg/L = micrograms per litre.

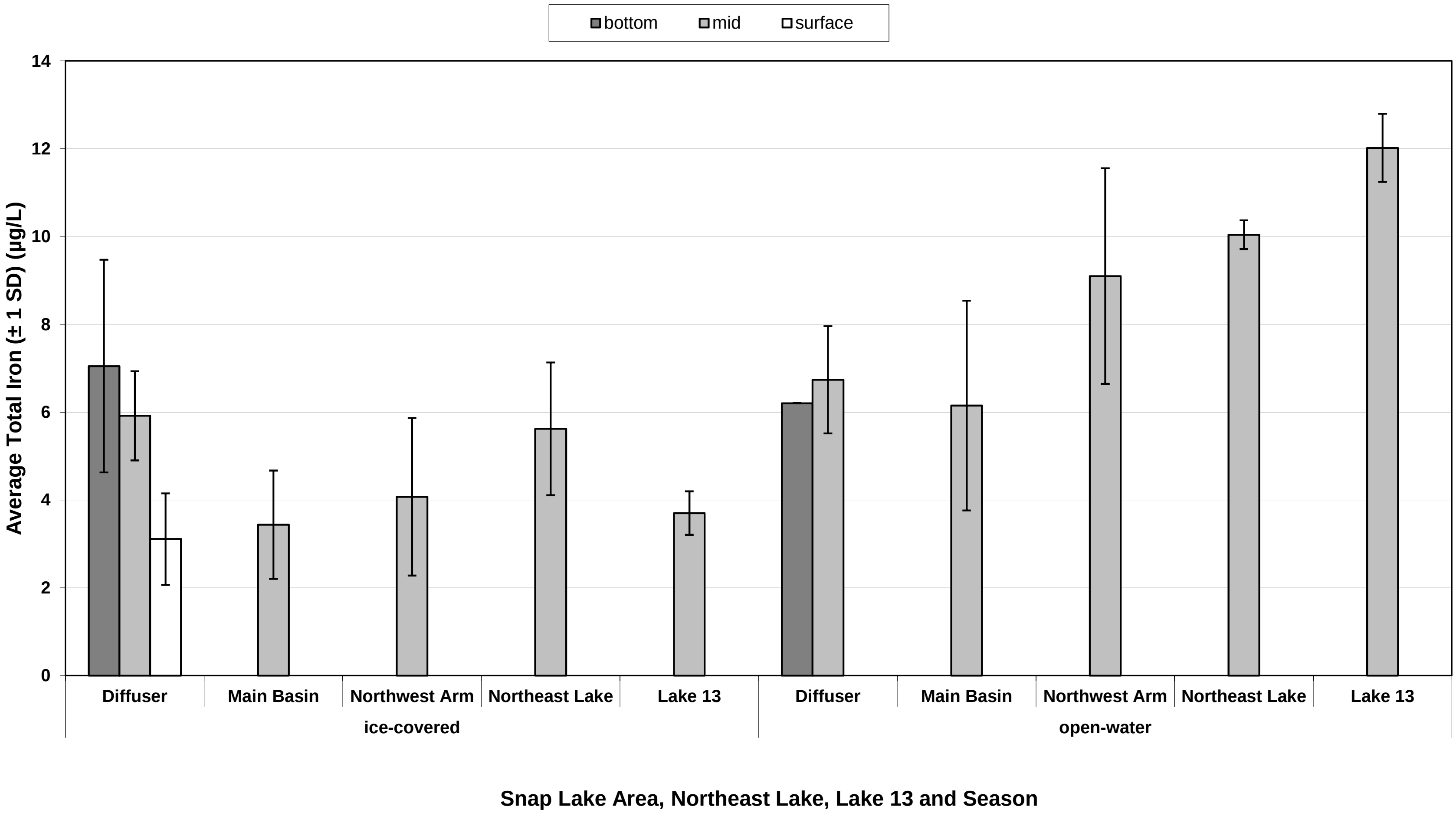
Figure 3-60 Average Total Antimony Concentrations (± 1 SD) in Snap Lake, Northeast Lake, and Lake 13, 2013



During the open-water season, the bottom average concentration for the diffuser stations represents the concentration of one sample collected from the bottom of the lake at SNP 02-20e station on July 9, 2013, where the highest conductivity was measured.

SD = standard deviation; µg/L = micrograms per litre.

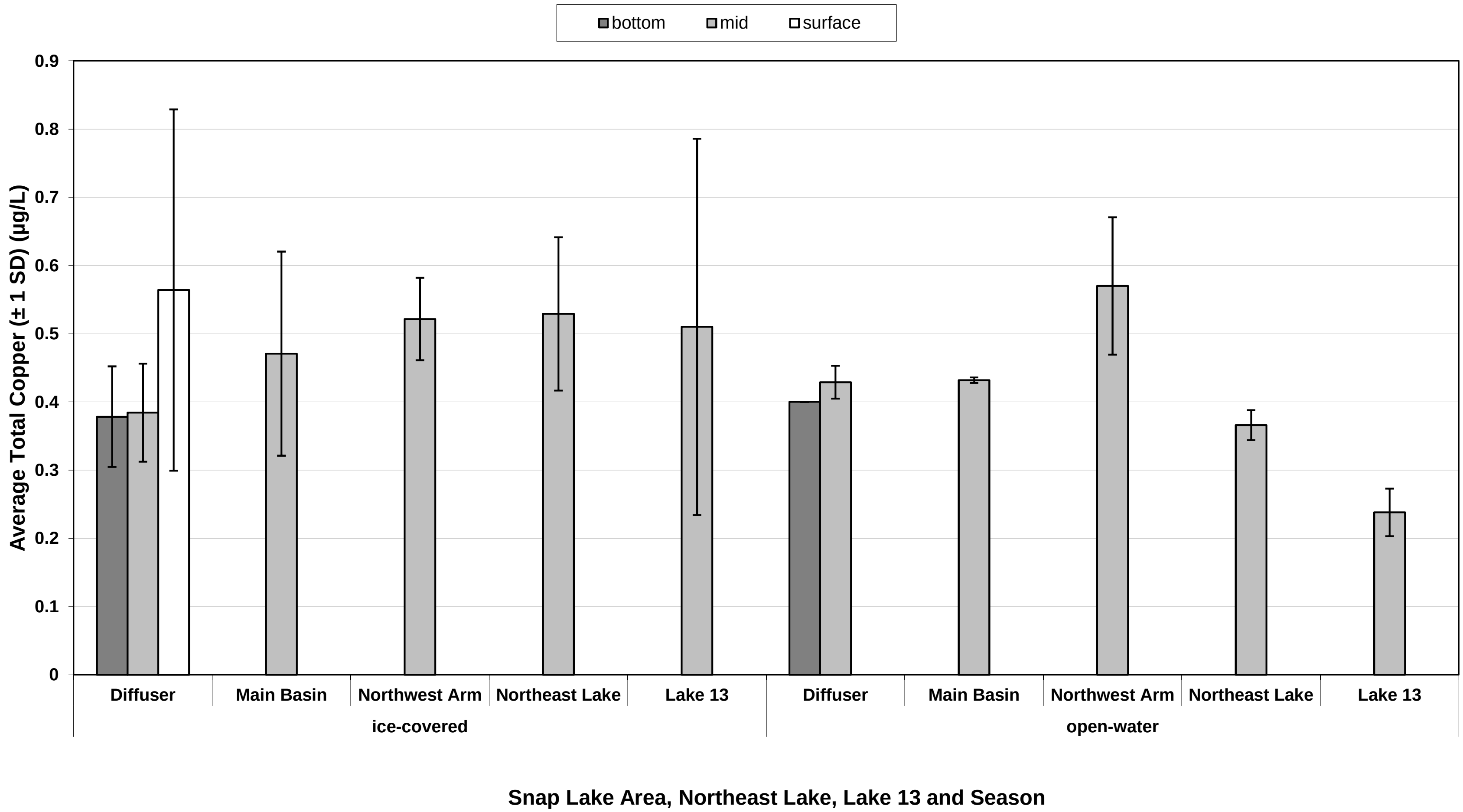
Figure 3-61 Average Total Iron Concentrations (± 1 SD) in Snap Lake, Northeast Lake, and Lake 13, 2013



During the open-water season, the bottom average concentration for the diffuser stations represents the concentration of one sample collected from the bottom of the lake at SNP 02-20e station on July 9, 2013, where the highest conductivity was measured.

SD = standard deviation; µg/L = micrograms per litre.

Figure 3-62 Average Total Copper Concentrations (± 1 SD) in Snap Lake, Northeast Lake, and Lake 13, 2013



During the open-water season, the bottom average concentration for the diffuser stations represents the concentration of one sample collected from the bottom of the lake at SNP 02-20e station on July 9, 2013, where the highest conductivity was measured.

SD = standard deviation; µg/L = micrograms per litre.

3.4.5.2 Water Quality Downstream of Snap Lake

Summary of the Downstream Lakes Special Study

A Downstream Lakes Special Study was conducted as part of the 2013 AEMP to document the spatial extent of treated effluent downstream of Snap Lake. A detailed description of monitoring methods, results, and updated modelling predictions is provided in Section 11.3.2; a brief summary of the water quality component is provided below.

Three lakes located immediately downstream of Snap Lake were surveyed during the 2013 Downstream Lakes Special Study, based on evidence of treated effluent in these lakes in 2011 and 2012 (De Beers 2012b, 2013b). Concentrations of TDS and other Mine-related parameters (e.g., conductivity, major ions, and nitrate) decreased with distance downstream from Snap Lake, consistent with EAR predictions (De Beers 2002). Higher concentrations were measured in 2013 at the lakes located immediately downstream of Snap Lake (DSL1, DSL2), and at Inlet of Lac Capot Blanc, when compared to those measured at the stations located farther in Lac Capot Blanc (Section 11.3.5). The same decreasing pattern was also observed in concentrations of total metals including barium, boron, lithium, molybdenum, nickel, rubidium, strontium, and uranium, which are also characteristic of the treated effluent. Field conductivity decreased from 285 µS/cm at the inlet of Lac Capot Blanc, to near background levels at the two outlets of Lac Capot Blanc (Section 11.3.5).

In 2013, the presence of treated effluent from Snap Lake was observed approximately 5 km away from the inlet of Lac Capot Blanc, approximately 11 km downstream from the Snap Lake outlet, and approximately 5 km farther downstream than 2012 (De Beers 2013b; Section 11.3). Consistent with EAR predictions, maximum TDS concentrations in lakes downstream of Lac Capot Blanc were predicted to decrease with distance downstream.

King Lake

Water quality at the furthest downstream AEMP station, KING01, which is located 25 km downstream of Snap Lake, was characterized by slightly acidic to neutral pH (6.7 to 7.3) and low alkalinity conditions (Table 3-17). Calculated TDS values ranged from 15 to 20 mg/L in 2013 (Figure 3-63).

Nutrient concentrations were undetectable or near DLs; consequently, all nutrient concentrations were below AEMP benchmarks. Metals concentrations were low and also below their AEMP benchmarks (Table 3-17).

A Seasonal Kendall test for temporal trend was conducted on TDS data collected at KING01. The results of the Seasonal Kendall test identified a significant upward trend in TDS concentrations at station KING01, with a P-value of 0.005 (Table 3-18), consistent with the results from 2012 (De Beers 2013b). A P-value of less than 0.05 indicates a significant trend in the data. The maximum TDS concentration at KING01 increased from 12 mg/L in 2005 to 20 mg/L in 2013, still considered to be within baseline levels for KING01 (De Beers 2002). Because station KING01 is located 25 km downstream of Snap Lake, additional volumes of low TDS concentration waters from the downstream watershed provide substantial dilution to inflows sourced from Snap Lake. In the EAR, concentrations were conservatively predicted to reach near background concentrations within 44 km of Snap Lake during the operating period when maximum concentrations in Snap Lake were predicted to occur.

Table 3-17 Comparison of Water Quality Results at the Downstream Station KING01 and Snap Lake

Parameter	Units	AEMP Benchmarks (Protection of Aquatic Life)[a]	Observed Concentrations [b]		
			Type	Snap Lake	Downstream Station (KING01)
Field Parameters					
pH	unitless	6.5 to 9.0	range	**5.3** to 8.0	**5.8** to 6.4
Conventional Parameters					
Laboratory pH	unitless	6.5 to 9.0	range	6.8 to 7.8	6.7 to 7.3
Total dissolved solids, calculated (Lab)[c]	mg/L	-	max	301	20
Ions					
Chloride	mg/L	120	max	**134**	3
Fluoride	mg/L	0.12	max	**0.23**	0.10
Sodium	mg/L	-	max	33	1
Calcium	mg/L	-	max	61	3
Magnesium	mg/L	-	max	7.7	1.0
Sulphate	mg/L	-	max	25	2
Alkalinity, as $CaCO_3$	mg/L	-	range	10 to 34	6 to 13
Hardness, as $CaCO_3$	mg/L	-	range	37 to 185	11 to 11
Nutrients					
Nitrate, as N	mg-N/L	2.93	max	**3.04**	<0.01
Nitrite, as N	mg-N/L	0.06	max	0.026	<0.002
Ammonia, as N	mg-N/L	0.51 to 125 [d]	max	0.29	0.01
Total phosphorus	mg-P/L	-	max	0.013	0.003
Total Metals					
Aluminum	µg/L	5 to100[e]	max	8	3
Arsenic	µg/L	5	max	0.1	0.1
Barium	µg/L	-	max	29	4
Boron	µg/L	1,500	max	69	5
Cadmium	µg/L	0.36	max	0.013	<0.005
Chromium	µg/L	8.9	max	0.17	0.08
Hexavalent chromium	µg/L	2.1	max	1.3	<1
Copper	µg/L	7.9	max	1.1	0.6
Iron	µg/L	300	max	12	4
Lead	µg/L	1 to 6.91[f]	max	0.03	<0.01
Lithium	µg/L	-	max	14	1
Manganese	µg/L	-	max	14	1
Mercury (Flett)	µg/L	0.026	max	0.002	<0.001

Table 3-17 Comparison of Water Quality Results at the Downstream Station KING01 and Snap Lake

Parameter	Units	AEMP Benchmarks (Protection of Aquatic Life)[a]	Observed Concentrations [b]		
			Type	Snap Lake	Downstream Station (KING01)
Total Metals (Continued)					
Molybdenum	µg/L	73	max	1.73	<0.05
Nickel	µg/L	25 to 150 [f]	max	2.64	0.38
Selenium	µg/L	1	max	0.1	<0.04
Silver	µg/L	0.1	max	0.006	<0.005
Thallium	µg/L	0.8	max	<0.01	<0.01
Uranium	µg/L	15	max	0.29	0.02
Zinc	µg/L	30	max	3	<1

a) AEMP benchmarks are: water quality guidelines (WQGs) from the Canadian Council of Ministers of the Environment (CCME) (1999) and site-specific EAR benchmarks developed for the protection of aquatic life for copper, chromium (VI) and cadmium (5% Probable Effect Level) from De Beers (2002). **Bold** values are above the relevant benchmarks.

b) Observed concentrations within the 2013 reporting period (November 1, 2013 to October 31, 2013).

c) Total dissolved solids calculated using formula adapted from Method 1030 E in Standard Methods for the Examination of Water and Wastewater, 21st Edition (APHA 2005).

d) The ammonia WQG is pH and water temperature dependent. The CCME recommended that the guideline values falling into the range less than 5 degrees Celsius (°C) and greater than pH of 10 should be used with caution because the lack of toxicity data to accurately determine the toxicity effects at high and low extremes. Therefore, the range of the guideline shown is based on a range of the maximum field pH (8.1) and temperature (16.9°C) in Snap Lake over the 2013 reporting period and the lowest pH (6.0) and temperature (5°C) recommended in (CCME 1999). The guideline was calculated based on an individual pH and water temperature for each sample with the final value expressed as ammonia nitrogen.

e) The aluminum WQG is pH dependent. The guideline shown is based on a range of field pH observed in Snap Lake during the 2013 reporting period (5.3 to 8.1). The WQG was calculated based on the individual pH for each sample.

f) Lead and nickel WQGs are hardness dependent. The range of the WQGs shown here was based on a range of hardness from 37 to 185 mg/L, which was observed in Snap Lake during the 2013 reporting period. The WQG was calculated based on the individual hardness for each sample.

Flett = Flett Research Limited; - = not applicable; N = nitrogen; $CaCO_3$ = calcium carbonate; max = maximum;
<= less than;% = percent; µg/L = microgram per litre; mg/L = milligram per litre; mg-N/L = milligrams as nitrogen per litre;
mg-P/L = milligrams as phosphorus per litre; °C = degrees Celsius; AEMP = Aquatic Effects Monitoring Program.

Table 3-18 Summary of Temporal Trend for Total Dissolved Solids at Station KING01 using the Seasonal Kendall Test

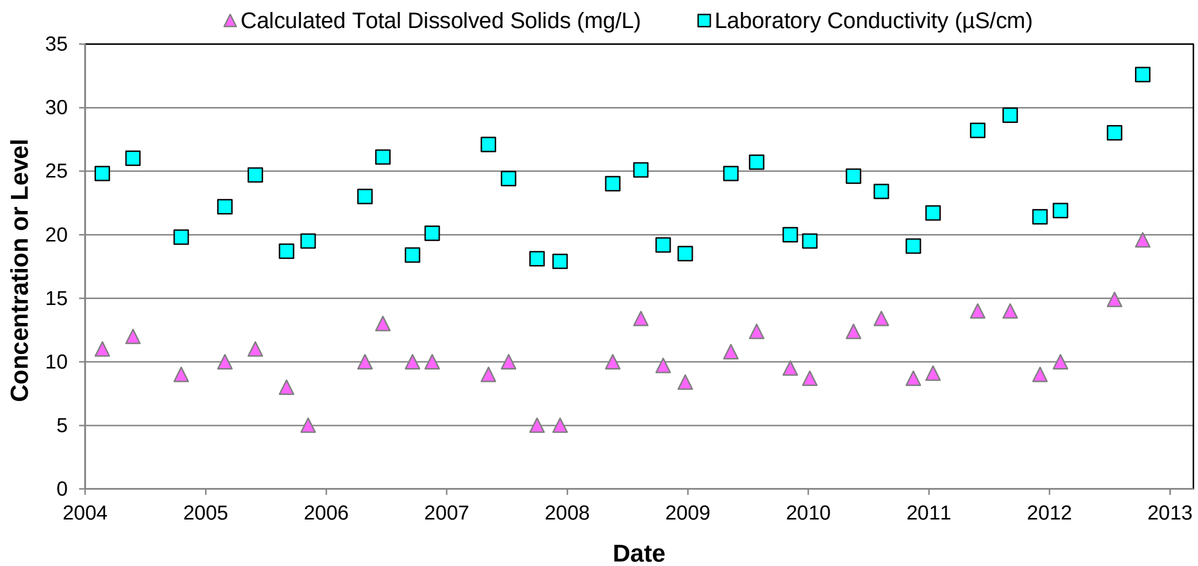
Parameter	Station	Depth	N	*Z*-Value at 95% Confidence[a]	*P*-Value at 95% Confidence[a]	Significant Trend
Total Dissolved Solids, Calculated (as a two-sided trend)	KING01	Mid	34	2.779	0.005	↑

Note: The Seasonal Kendall Test was run using SYSTAT 13.00.05 (SYSTAT 2009).

a) The critical *Z*-values associated with a two-sided 95% confidence interval are -1.96 and 1.96. The *P*-value associated with a 95% confidence interval is 0.05. If the *Z*-value is between -1.96 and 1.96 for a two-sided trend test, the P-value will be greater than 0.05 and the test concludes that no significant increasing or decreasing trend exists in the data.

↑ = an increasing trend; % = percent; n = sample count.

Figure 3-63 Concentrations of Total Dissolved Solids and Conductivity at the Downstream Station KING01, 2005 to 2013



mg/L= milligrams per litre; µS/cm = microSiemens per centimetre.

3.4.5.3 Summary of Key Question 4

Spatial and seasonal patterns were observed for some water quality parameters in Snap Lake. Concentrations of TDS and water quality parameters related to Mine activity (conductivity, major ions, nitrogen-nutrients [nitrate, nitrite, ammonia, TKN, TN], and eight metals that correlated with conductivity [barium, boron, lithium, molybdenum, nickel, rubidium, strontium, and uranium]) were higher in the main basin relative to the northwest arm, as well as higher in all areas of Snap Lake relative to reference lakes. Spatial gradients within the main basin of Snap Lake for these parameters were less prominent in 2013 compared to gradients observed in the first four years of treated effluent discharges to Snap Lake (i.e., 2004 to 2007). Concentration differences in Mine-related parameters were lower in the main basin of Snap Lake relative to differences observed in the northwest arm.

Concentrations of most treated effluent-related parameters in the northwest arm continue to be notably lower compared to the main basin due to the limited hydraulic connection between the northwest arm and the main basin. However, the concentrations observed in the northwest arm were higher than those observed at Northeast Lake and Lake 13, and the increased concentration was evident close to the northwest arm's narrow connection to the main basin of Snap Lake in 2013.

Manganese and cobalt concentrations were higher in Snap Lake relative to the reference lakes in 2013. Although these two metals are at similar concentrations in the treated effluent as barium and uranium, manganese and cobalt were not positively correlated with conductivity, which is typically used as an indicator of treated effluent, and had no clear spatial pattern within Snap Lake. These metals may be behaving less conservatively than other effluent-related parameters due to oxidation and reduction processes, controlled partly by DO concentrations, in Snap Lake.

The lack of a clear spatial patterns in phosphorus was consistent with previous spatial assessments and likely indicates that either phosphorus removal processes are reducing phosphorus concentrations in the water column (i.e., through phytoplankton uptake and/or sedimentation processes) or differences are too subtle to observe, given the uncertainty in low level phosphorus measurements.

Vertical gradients in conductivity are affected by the discharge of treated effluent, and also likely by thermoclines that occur at deeper stations during open-water conditions. During ice-covered conditions, vertical gradients in conductivity showed that the treated effluent plume near the diffuser continued to sit near the middle of the water column. The diffuser discharges treated effluent away from the lake bottom into the water column; the treated effluent appeared not to sink or rise, likely due to the similar densities between the treated effluent and lake waters. During open-water conditions, conductivity gradients did not occur at most stations, likely due to wind-driven mixing. Gradients during open-water conditions were still observed at deeper stations, particularly at the beginning of the open-water season when wind-driven mixing may not have yet influenced the lower portion of the water column. In addition, thermoclines were observed at deeper stations in Snap Lake during open-water conditions (in addition to both reference lakes); the large density differences caused by such sharp decreases in temperatures differences with depth may have also inhibited mixing at these locations.

Vertical gradients in DO near the diffuser are also likely affected by the discharge of treated effluent, whereby the well-oxygenated treated effluent increases DO near the diffuser. At most stations during ice-covered conditions and at deeper stations in open-water conditions, vertical gradients of decreasing DO concentrations with depth occurred in Snap Lake and the reference lakes. However, at stations near the diffuser during ice-covered conditions and at the deepest diffuser station during open-water conditions, the DO concentrations did not decrease to the same degree.

Seasonal differences between ice-covered and open-water conditions in effluent-related parameters were less prominent in 2013 compared to the more pronounced differences observed between 2004 and 2007. The reduction in the range of seasonal and spatial differences is attributed to the greater exposure of treated effluent discharge within Snap Lake.

Results from the Downstream Lakes Special Study (Section 11.3) showed evidence of an influence of the treated effluent throughout lakes DSL1 and DSL2, and near the inlet of Lac Capot Blanc in 2013. Concentrations of Mine-related constituents reached background concentrations approximately 11 km downstream of Snap Lake, which is approximately 5 km further downstream than observed in 2012. At 25 km downstream (at KING01), an increasing trend in TDS concentrations was confirmed, but TDS concentrations remained within baseline levels near King Lake.

3.4.6 Key Question 5: Is there Evidence of Acidification Effects from the Mine on Nearby Waterbodies?

As part of the 2009 AEMP acidification assessment (De Beers 2010), additional monitoring of inland lakes was recommended because the potential for contribution of the Mine to acidification of inland lakes 3 and 4 (IL3 and IL4) could not be ruled out at that time. The following section includes a summary and qualitative review of water quality data collected during 2013 for three inland lakes (IL3, IL4, and IL5), the two major tributaries to Snap Lake, and Streams S1 and S27, to identify sensitivity to acidification and any changes resulting from potentially acidifying depositions from Mine emissions.

3.4.6.1 Inland Lakes

The three inland lakes monitored in 2013 were characterized by low TDS, neutral to slightly acidic pH, and low alkalinity (Table 3-19), consistent with the water quality observed in 2012 (De Beers 2013a). Dissolved oxygen concentrations measured in the inland lakes were within WQGs for the protection of aquatic life. Two laboratory pH values measured in lakes IL3 and IL4 were just below the WQG range. Concentrations of major ions and nitrogen parameters in the inland lakes were well below the CCME WQGs (Table 3-19). The 2013 ammonia concentrations in all three inland lakes remained below WQGs (CCME 1999). Using the classification system of Saffran and Trew (1996), alkalinity concentrations indicate that all three inland lakes are highly sensitive to acidification (i.e., total alkalinity was less than or equal to 10 mg/L as $CaCO_3$).

Table 3-19 Summary of Selected Water Quality Results for Three Inland Lakes

Parameter Name	Unit	AEMP Benchmarks (Protection of Aquatic Life) [a]	Observed Concentrations[b]			
			Type	IL3	IL4	IL5
Field Parameter						
pH	unitless	6.5 to 9.0	range	6.7 to 7.4	6.9 to 7.5	6.5 to 7.1
Dissolved Oxygen	mg/L	6.5, 9.5[c]	min, mean	6.7, 8.6	6.8, 9.0	8.4, 9.5
Specific Conductivity	µS/cm	-	mean	21	23	44
Conventional Parameters						
Laboratory pH	unitless	6.5 to 9.0	range	**6.5** to 6.7	**6.4** to 6.6	6.9 to 7.1
Total Dissolved Solids, calculated (Lab)	mg/L	-	mean	8.4	9.2	22
Laboratory Specific Conductivity	µS/cm	-	mean	21	22	44
Major Ions						
Chloride	mg/L	120	mean	<0.5	0.6	1.4
Fluoride	mg/L	0.12	mean	0.05	0.05	0.05
Hardness, as $CaCO_3$	mg/L	-	mean	7.4	7.9	15.6
Sodium	mg/L	-	mean	1.0	1.1	1.6
Sulphate	mg/L	-	mean	1.2	0.5	6.1
Total Alkalinity, as $CaCO_3$	mg/L	-	mean	4.6	4.7	10

Table 3-19 Summary of Selected Water Quality Results for Three Inland Lakes

Parameter Name	Unit	AEMP Benchmarks (Protection of Aquatic Life) [(a)]	Observed Concentrations[(b)]			
			Type	IL3	IL4	IL5
Nutrients						
Nitrate, as N, calculated	mg-N/L	2.93	mean	0.02	<0.006	<0.006
Nitrite, as N	mg-N/L	0.06	mean	0.003	0.005	<0.002
Total Ammonia, as N	mg-N/L	0.9 to 39.7[(d)]	mean	0.03	0.06	0.02

Bold values are above the relevant water quality guidelines (WQGs) or outside the guideline range for pH. Values were rounded to reflect laboratory or field instrument precision after comparisons to guidelines. Therefore, values slightly above guidelines (e.g., laboratory pH in IL3 and IL 4) were displayed as being equal to the guidelines buy were identified as exceedances. Measured concentrations equal to the guideline values (e.g., field pH in IL5) were not identified as exceedances.

a) Water Quality Guidelines (WQGs) are from the Canadian Council of Ministers of the Environment (CCME) (1999)

b) Observed concentrations during the 2013 inland lake sampling period (July to September 2013).

c) Lowest acceptable dissolved oxygen concentration for cold-water biota is 9.5 mg/L for early life stages; 6.5 mg/L for other life stages.

d) Ammonia WQG is pH and water temperature dependent. Range of the WQG shown is based on a range of field pH from 6.5 to 7.5 and a range of water temperature from 8.8°C to 24.8°C, which were observed in the inland lakes during the 2013 inland lake sampling period (July to September, 2013). The WQG was calculated based on an individual field pH and water temperature for each sample with the final value expressed as ammonia as nitrogen.

- = not applicable; µS/cm = microSiemens per centimetre; $CaCO_3$ = calcium carbonate; N = nitrogen; mg/L = milligrams per litre; mg-N/L = milligrams as nitrogen per litre; < = less than; min = minimum.

Comparison to Key Acidification Indicator Parameters

Sulphate concentrations in IL3 and IL4 in 2013 were lower than baseline concentrations (i.e., concentrations measured in 1999 and 2002) (Figure 3-64). Overall, concentrations of sulphate in IL3 and IL4 have remained relatively stable since annual monitoring of the inland lakes began in 2007. Sulphate concentrations in IL5, although elevated compared to baseline, are still within the range of concentrations observed in the other inland lakes during baseline conditions. Elevated concentrations of sulphate observed in IL5 since 1999 could potentially indicate increased sulphate loadings as a result of Mine emissions.

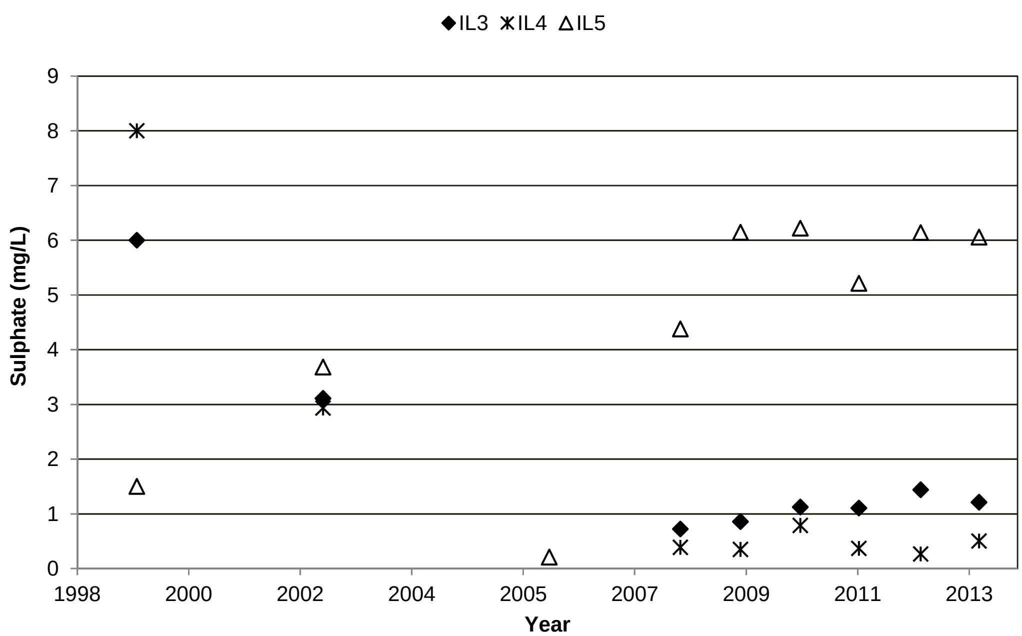
Nitrate concentrations in the inland lakes in 2013 were below detection limits and were near baseline concentrations (Figure 3-65). Concentrations of nitrate were elevated in IL5 between 2006 and 2009, but were much lower from 2010 to 2013.

Base cation concentrations were consistently higher than baseline concentrations in the inland lakes over the 13 year sampling record (Figure 3-66), and have remained relatively stable since annual monitoring of the inland lakes began in 2007. The noted increase in base cations from 2006 to 2008, compared to baseline concentrations, could indicate leaching from soils into surrounding catchment, or conversely, could be indicative of increased weathering or deposition (UNECE 2004).

Measurements of total alkalinity in 2013 were comparable to those observed between 2008 and 2012 in the inland lakes (Figure 3-67). Laboratory pH values have consistently decreased since 2010 in all three lakes (Figure 3-68). Field measured pH values varied within a similar range to the laboratory pH values across the sampling years, but did not show a consistently decreasing trend (Figure 3-69). The temporal trend differences between the laboratory and field pH values are likely related to differences in field and laboratory pH values discussed in the QA/QC assessment (Appendix 3A). Both the 2013 laboratory and field pH values in the inland lakes remained higher or within the range in baseline pH values measured in the laboratory and field, respectively.

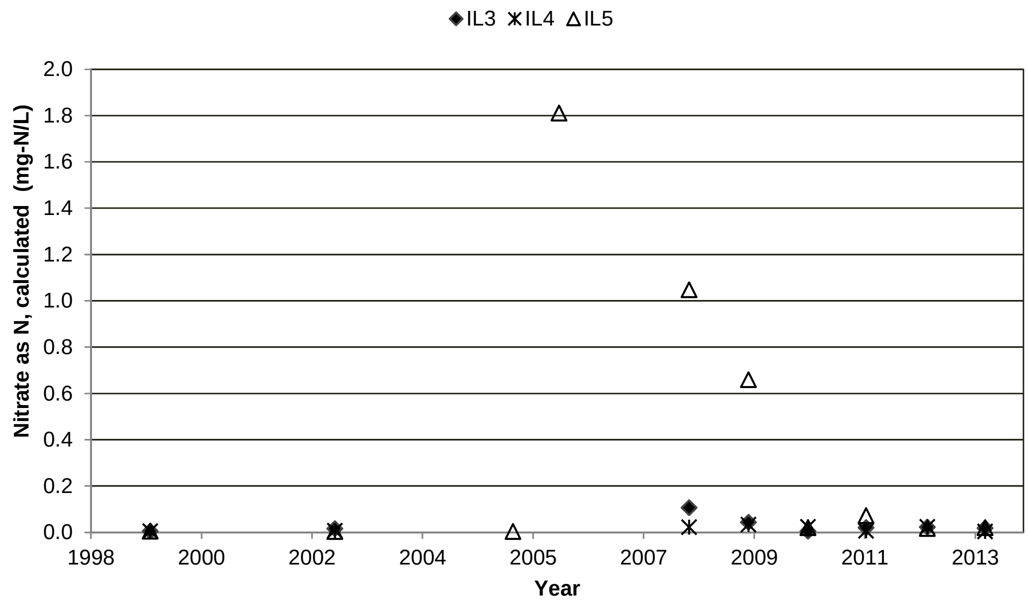
Based on the 2013 data, there was no strong evidence of acidification of these lakes. These results are consistent with previous assessments (De Beers 2010, 2011, 2012b, 2013a), which concluded that there is limited potential for acidification of these lakes due to emissions from the Mine. However, data shown in Figures 3-63 to 3-67 indicate that water quality in these lakes may be changing over time. These changes are particularly noticeable at IL5, where concentrations of sulphate, base cations, and alkalinity are elevated compared to baseline. Within lake fluctuations of sulphate, nitrate, base cations, and alkalinity observed in IL3 and IL4 appear to be within the normal range of variation seen since monitoring began in 2007, but are generally different from baseline conditions. The 2013 laboratory pH values and field pH values measured in the inland lakes remained higher, or within the range in baseline pH values measured in the laboratory and field, respectively.

Figure 3-64 Sulphate Concentrations in Inland Lakes 3, 4, and 5, 1999 to 2013



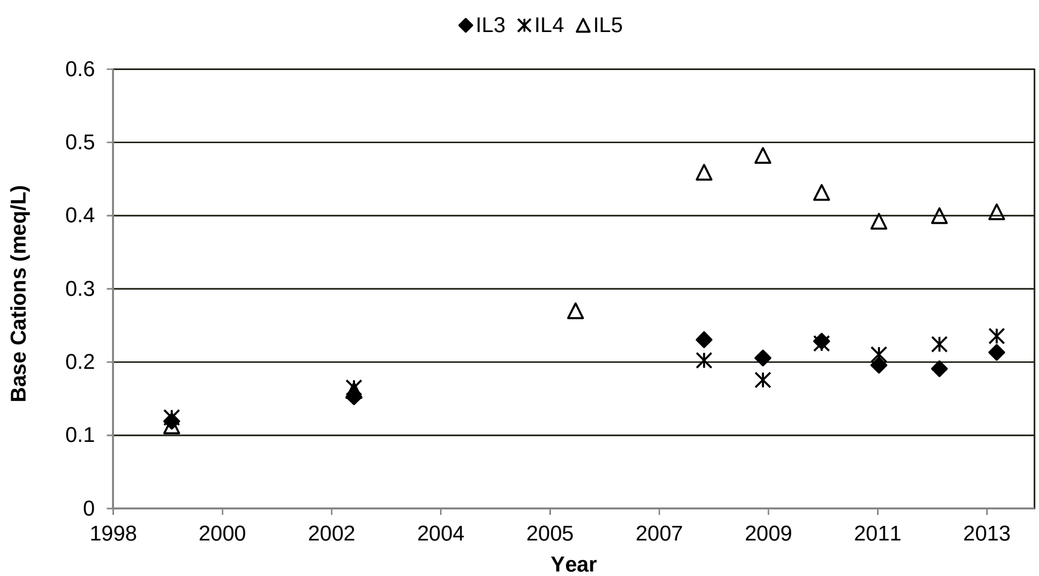
Note: Mean concentrations are shown for 2008 to 2013; values prior to 2008 represent one discrete sample.
IL = inland lake; mg/L = milligram per litre

Figure 3-65 Nitrate Concentrations in Inland Lakes 3, 4, and 5, 1999 to 2013



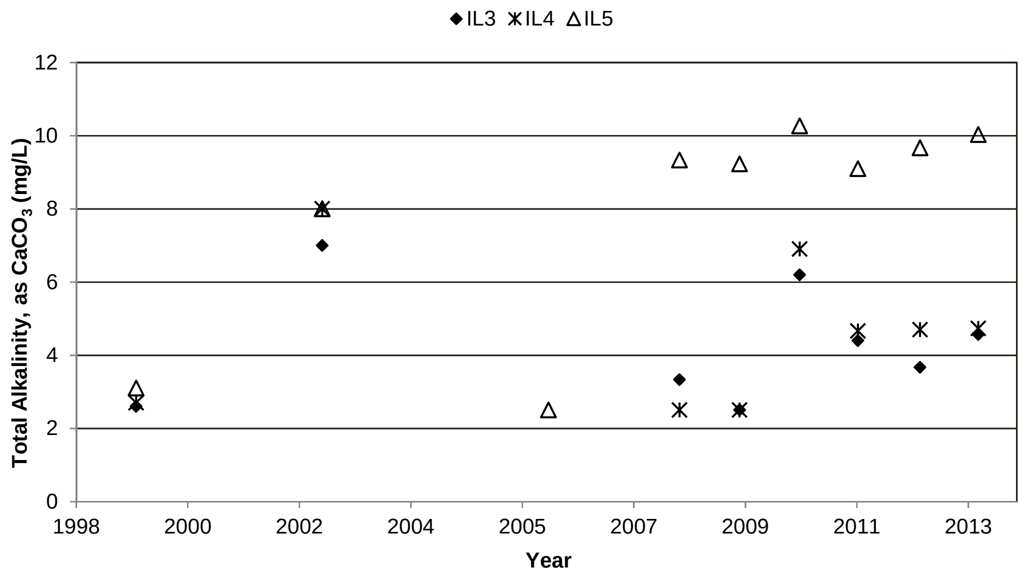
Note: Nitrate+ nitrite concentrations were plotted for the 1999 data, because nitrate values were not available.
Mean concentrations are shown for 2008 to 2013; values prior to 2008 represent one discrete sample.
IL= inland lake; N = nitrogen; mg-N/L= milligram as nitrogen per litre.

Figure 3-66 Sum of Base Cations in Inland Lakes 3, 4, and 5, 1999 to 2013



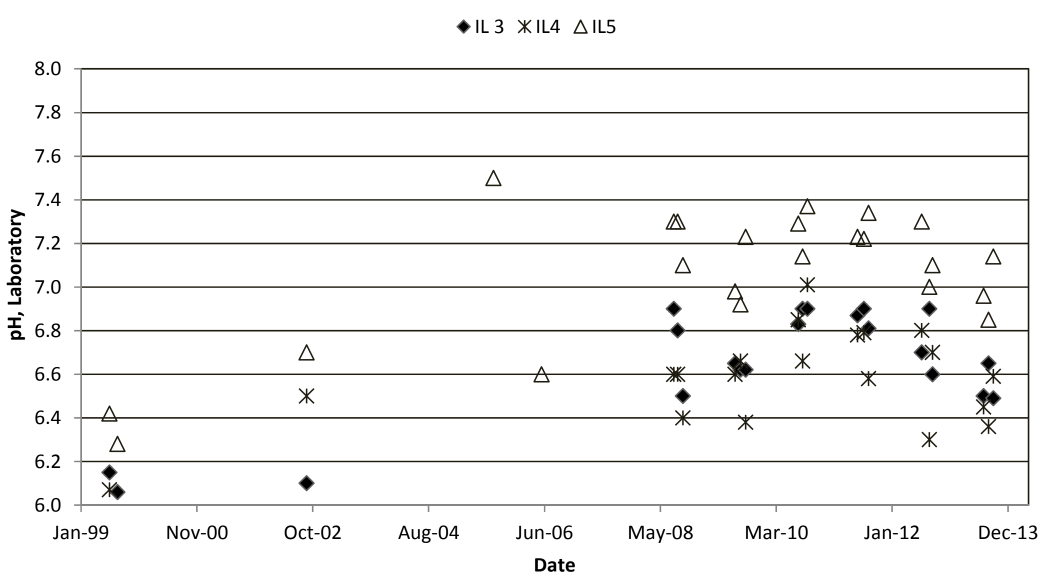
Note: Mean concentrations are shown for 2008 to 2013; values prior to 2008 represent one discrete sample.
IL= inland lake; meq/L = milliequivalent per litre.

Figure 3-67 Total Alkalinity in Inland Lakes 3, 4, and 5, 1999 to 2013



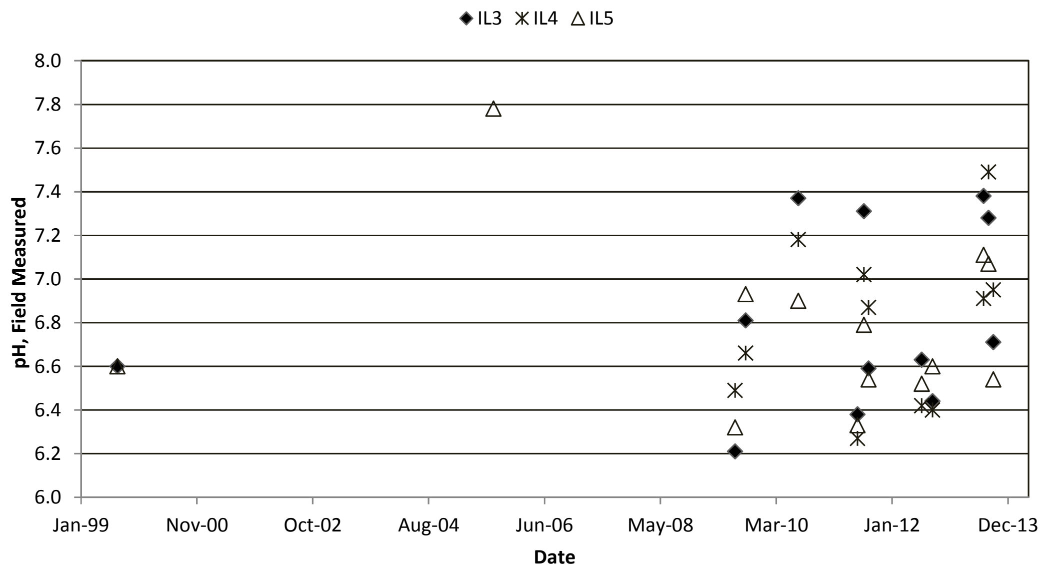
Note: Mean concentrations are shown for 2008 to 2013; values prior to 2008 represent one discrete sample.
IL= inland lake; $CaCO_3$ = calcium carbonate; mg/L = milligrams per litre.

Figure 3-68 Laboratory pH Values in Inland Lakes 3, 4, and 5, 1999 to 2013



IL = inland lake.

Figure 3-69 Field Measured pH Values in Inland Lakes 3, 4, and 5, 1999 to 2013



IL = inland lake.

3.4.6.2 Streams S1 and S27

Streams S1 and S27 are the two major tributaries flowing into Snap Lake (Figure 3-1). Stream S1 has been monitored since 2005 during spring freshet and open-water conditions as part of the AEMP. Water quality monitoring of Stream S27 has been completed sporadically since 1999 (i.e., years 1999, 2002, 2005, 2011, 2012 and 2013). Streams S1 and S27 are monitored to identify any changes in stream water quality related to mining activities, to establish loadings to Snap Lake from this input source (i.e., to support modelling and mass/water balance assessments), as well as to meet the EAR commitment to monitor potential spring acid pulses in Streams S1 and S27 (De Beers 2002).

Water quality in Streams S1 and S27 was characterized by low to moderate alkalinity, low concentrations of major ions, and slightly acidic waters. Aluminum and iron have typically been higher in Stream S1 (De Beers 2002, 2006, 2007a, 2008a, 2009, 2010, 2011, 2012b, 2013a) and Stream S27 (De Beers 2002, 2013a) compared to Snap Lake. In 2013, the maximum concentrations of all water quality parameters measured from Streams S1 and S27 were below AEMP benchmarks with three exceptions (Table 3-20). The minimum field measured and laboratory pH values in Stream S1 were measured below the WQG, but were within the normal pH range for field and laboratory measured values, respectively (Figures 3-70 and 3-71). The minimum field pH value was measured below the WQG but was within the normal pH range in Stream S27. Maximum aluminum and iron concentrations were above the WQG in both streams. Aluminum was also above the WQG in the baseline (i.e., 1999) metals data collected from S27 (De Beers 2002). Iron concentrations have typically been elevated in Streams S1 and S27 in all sampling years since the baseline (i.e., 2005, 2011 to 2013).

The ranges in field measured pH, laboratory pH, total alkalinity, and sulphate values in Streams S1 and S27 in 2013 were similar to baseline values (Figures 3-70 to 3-73) presented in Appendix IX.6 of the EAR (De Beers 2002); therefore, no evidence of acidification, or any spring acid pulse, was discernible in Streams S1 and S27 in 2013.

Table 3-20 Summary of Selected Water Quality Results for Streams S1 and S27, 2013

Parameter	Units	AEMP Benchmarks (Protection of Aquatic Life)[a]	Observed Concentrations[b]		
			Type	Stream S1	Stream S27
Field Parameters					
pH	unitless	6.5 to 9.0	range	**6.0** to 7.3	**6.2** to 6.8
Dissolved Oxygen	mg/L	6.5, 9.5[c]	min	7.59	6.99
Conventional Parameters					
Laboratory pH	unitless	6.5 to 9.0	range	**6.4** to 7.1	6.7 to 6.9
Total Dissolved Solids, calculated (Lab)	mg/L	-	max	16	14
Major Ions					
Calcium	mg/L	-	max	3.0	2.5
Chloride	mg/L	120	max	1.0	0.5
Fluoride	mg/L	0.12	max	0.05	0.04
Hardness, as CaCO3	mg/L	-	range	6.7 to 12	7.9 to 11
Magnesium	mg/L	-	max	1.0	1.3
Sodium	mg/L	-	max	1.3	1.3
Sulphate	mg/L	-	max	1.7	0.7
Total Alkalinity, as CaCO3	mg/L	-	range	<5 to 10	5.5 to 11
Nutrients					
Nitrate, as N	mg-N/L	2.93	max	0.02	0.17
Nitrite, as N	mg-N/L	0.06	max	0.005	0.003
Total Ammonia, as N	mg-N/L	4.0 to 125[d]	max	0.04	0.06
Total Metals					
Aluminum	µg/L	100[e]	max	**138**	**120**
Arsenic	µg/L	5	max	0.1	0.1
Boron	µg/L	1500	max	5.9	6.5
Cadmium	µg/L	0.36	max	0.03	0.02
Chromium	µg/L	8.9	max	0.3	0.2
Copper	µg/L	7.9	max	2.1	1.4
Iron	µg/L	300	max	**560**	**806**
Lead	µg/L	1.0 to 6.9[f]	max	0.03	0.03
Mercury[g]	µg/L	0.026	max	0.002	<0.02
Molybdenum	µg/L	73	max	0.3	0.1
Nickel	µg/L	45 to 152	max	1.6	0.5
Selenium	µg/L	1	max	0.05	<0.04
Silver	µg/L	0.1	max	<0.005	<0.005

Table 3-20 Summary of Selected Water Quality Results for Streams S1 and S27, 2013

Parameter	Units	AEMP Benchmarks (Protection of Aquatic Life)[a]	Observed Concentrations[b]		
			Type	Stream S1	Stream S27
Total Metals					
Thallium	µg/L	0.8	max	<0.01	<0.01
Uranium	µg/L	15	max	0.05	0.05
Zinc	µg/L	30	max	3.9	4.6

Bold values are above or out of the acceptable range of AEMP benchmarks.

a) Water Quality Guidelines (WQGs) are from the Canadian Council of Ministers of the Environment (CCME) (1999) and EAR benchmarks for protection of aquatic life for copper, chromium (VI) and cadmium (5% Probable Effect Level) are from De Beers (2002).

b) Observed concentrations during the 2013 sampling period (May to September 2013).

c) Lowest acceptable dissolved oxygen concentration for cold-water biota is 9.5 mg/L for early life stages; 6.5 mg/L for other life stages.

d) The ammonia WQG is pH and water temperature dependent. The CCME recommended that the guideline values falling into the range less than 5 degrees Celsius (°C) and greater than pH of 10 should be used with caution because the lack of toxicity data to accurately determine the toxicity effects at high and low extremes. Therefore, the range of the guideline shown is based on a range of the maximum field pH (8.1) and temperature (16.9°C) in Snap Lake over the 2013 reporting period and the lowest pH (6.0) and temperature (5°C) recommended in the ammonia guideline (CCME 1999). The guideline was calculated based on an individual pH and water temperature for each sample with the final value expressed as ammonia nitrogen.

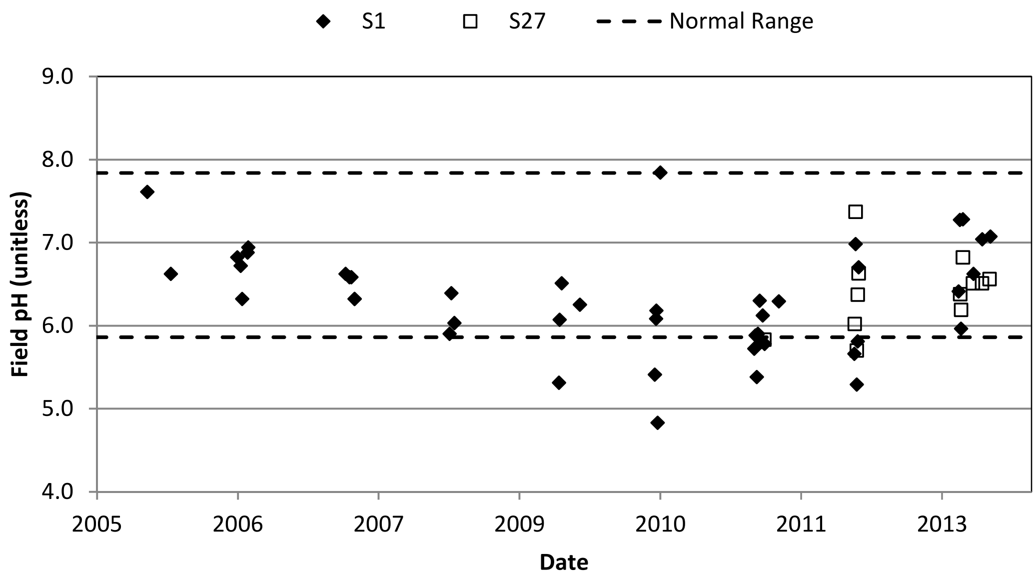
e) Aluminum WQG is pH dependent. The WQG shown here is based on a range of pH from 6.0 to 7.3, which was observed in Streams 1 and 27 during the 2013 reporting periods. The WQG was calculated based on the individual pH for each sample.

f) Lead and nickel WQGs are hardness dependent. The range of the WQGs shown here was based on a range of hardness from 6.7 to 11.6 mg/L, which was observed in Streams 1 and 27 during the 2013 reporting period. The WQG was calculated based on the individual hardness for each sample.

g) Mercury results analyzed by Flett Research Limited were included for Streams S1 and S27.

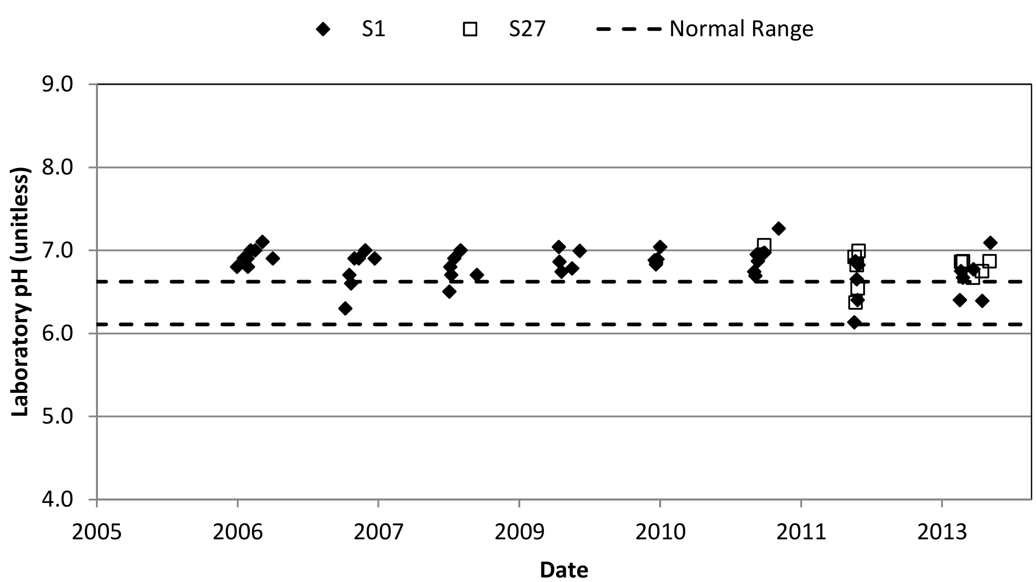
- = not applicable; $CaCO_3$ = calcium carbonate; N = nitrogen; < = less than; max = maximum; min = minimum; µg/L = micrograms per litre; mg/L = milligrams per litre; mg-N/L = milligrams as nitrogen per litre.

Figure 3-70 Field Measured pH in Stream 1 and Stream 27, 2005 to 2013



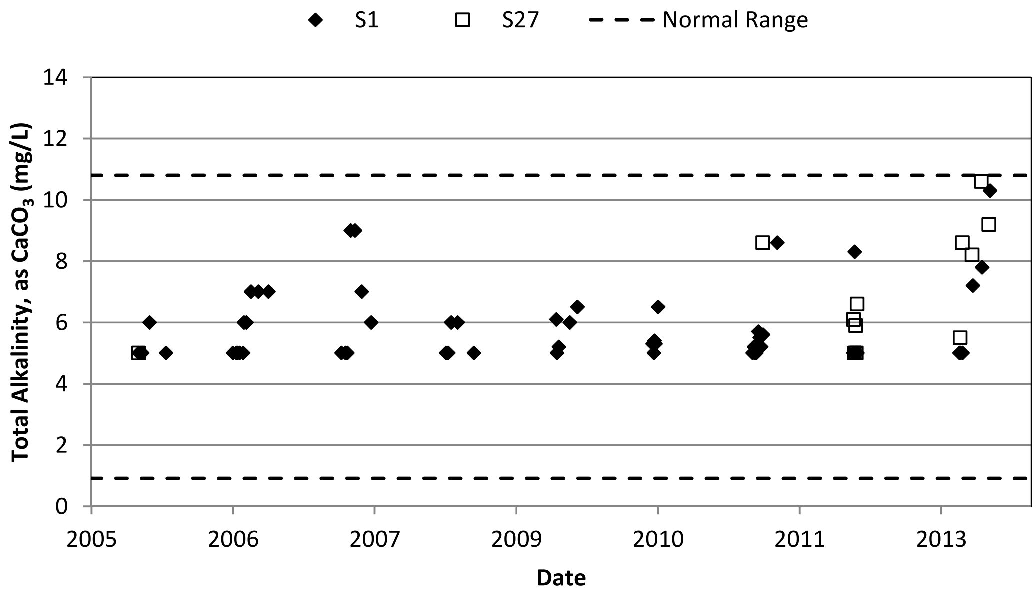
Note: Normal range based on data collected in 2001, with the upper and lower range calculated as the mean ± 2 SDs.
S1 = Stream S1; S27 = Stream S27; SD = standard deviation.

Figure 3-71 Concentrations of Laboratory pH in Stream 1 and Stream 27, 2005 to 2013



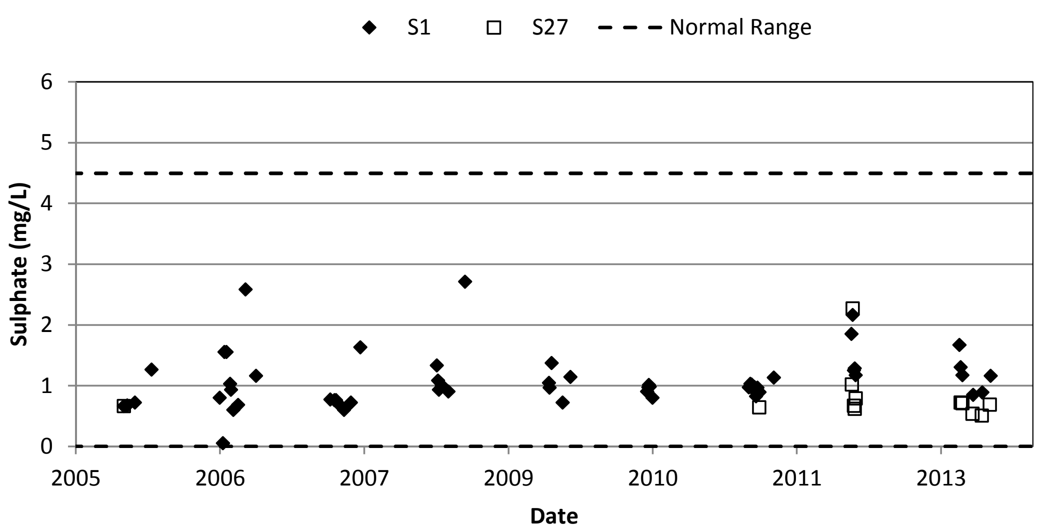
Note: Normal range based on data collected from 1999 to 2002, with the upper and lower range calculated as the mean ± 2 SDs.
S1 = Stream S1; S27 = Stream S27; SD = standard deviation.

Figure 3-72 Concentrations of Total Alkalinity in Stream 1 and Stream 27, 2005 to 2013



Note: Normal range based on data collected from 1999 to 2002, with the upper and lower range calculated as the mean ± 2 SDs.
S1 = Stream S1; S27 = Stream S27; $CaCO_3$ = calcium carbonate; mg/L = milligrams per litre; SD = standard deviation.

Figure 3-73 Concentrations of Total Sulphate in Stream 1 and Stream 27, 2005 to 2013



Note: Normal range based on data collected from 1999 to 2002, with the upper and lower range calculated as the mean ± 2 SDs.
S1 = Stream S1; S27 = Stream S27; mg/L = milligrams per litre; SD = standard deviation.

3.4.6.3 Summary of Key Question 5

Based on the 2013 data, there was no strong evidence of acidification of inland lakes IL3, IL4, and IL5 or of Streams S1 and S27. Although water quality in the inland lakes may be changing over time, results are consistent with previous assessments (De Beers 2010, 2011, 2012b, 2013a), which concluded that there is limited potential for acidification of these lakes due to emissions from the Mine. Monitoring of these lakes should be continued to confirm whether water quality is changing, particularly whether pH values in these lakes are decreasing below the range in baseline values. A decrease of pH below baseline values may require follow-up work to the acidification assessment completed in the EAR and in 2009 to determine whether the pH decreases are related to the Mine and what effects these changes could have on aquatic life.

The ranges in pH, total alkalinity, and sulphate values in Streams S1 and S27 in 2013 were similar to baseline values; therefore, no evidence of acidification, or any spring acid pulse, was discernible in Streams S1 and S27 in 2013.

3.4.7 Key Question 6: Is Water from Snap Lake Safe to Drink?

Water quality data collected from Snap Lake were compared to Canadian drinking WQGs for the aesthetic objective (AO) and maximum acceptable concentration (MAC; Table 3-21). Drinking WQGs for human consumption are more stringent than wildlife WQGs (CCME 1999; livestock watering guidelines). Therefore, water considered safe for humans to drink would also be adequate for wildlife consumption. If results were above a MAC and/or AO, an attempt was made to determine the relevance of the elevated results to potential for risk to human health and palatability. Where appropriate, this analysis involved additional comparison to average conditions and baseline conditions in Snap Lake, reviewing water quality at the water intake station (SNP 02-15), consideration of the frequency, duration and location of the elevated result, treatment practices, and the potential source of that parameter.

The assessment of Action Levels related to protecting drinking water in Snap Lake was also evaluated using 75% of relevant drinking water guidelines and wildlife health guidelines (i.e., livestock watering guidelines), which includes all AOs and MACs, with the exception of microbiological MACs (Table 3-21).

Concentrations of water quality parameters in samples collected from Snap Lake were below MACs and AOs (Health Canada 2012), with the exception of *E. coli* and total coliforms (Table 3-21). Microcystin-LR values in 2013 were below the DL and the drinking WQG (Appendix 3D, Table 3D-9). Iron was above the AO in one sample collected at the water intake location, SNP 02-15, in January 2013. *E. coli*, total coliforms and iron are discussed in more detail in the following two sections.

***Escherichia coli* and Total Coliforms**

In Snap Lake, the maximum *E. coli* value of 2 MPN/100 mL was above the MAC of no detectable *E. coli* per 100 mL (Table 3-21). The maximum was measured in one sample collected at mid-depth at diffuser station SNP 02-20f on July 9, 2013. However, *E. coli* were not detected in the remaining water samples collected during the 2013 AEMP (Appendix 3D; Table 3D-1). At the raw water intake from Snap Lake (i.e., station SNP 02-15, before water is filtered and chlorinated for drinking water purposes) one sample, collected on June 3, 2013, had an *E. coli* value of 1 MPN/100mL. However, this was the case with the 2013 AEMP samples; *E. coli* was not detected in the remaining water samples collected at SNP 02-15 during 2013 (Appendix 3I, Table 3I-1).

The maximum total coliform value of 411 CFU/100 mL, which was above the MAC of no detectable coliforms per 100 mL, was measured in the sample collected at SNP 02-15 on September 2, 2013 (Appendix 3I; Table 3I-1). Total coliforms were also detected near the water intake in the northwest arm during baseline (De Beers 2002). Total coliforms are commonly present in both surface water and groundwater from both human and non-human sources (Health Canada 2012). Results indicate that water from Snap Lake should be disinfected before human consumption, consistent with Health Canada's recommendation for all surface water in Canada (Health Canada 2012), because microbiological parameters can naturally exist in the aquatic environment. Currently, raw water pumped from Snap Lake is treated in the sequence of:

1. filtration through cartridge filters that consist of 5-µm filter and 0.35-µm filter in-line units; and,
2. chlorination.

The water is then tested for turbidity and chlorine at the temporary potable water treatment plant prior to public water use. Treated water is also tested for select microbiological parameters (*E. coli* and total coliforms) weekly (De Beers 2013a). If microbiology parameters are detected, all potable water tanks would be emptied and filters and tanks would be cleaned. Water would then be re-tested for microbiology parameters prior to public water use. The current water treatment at the Mine is designed and operated so that water consumed at the Snap Lake camp is acceptable for drinking from a microbiological perspective.

Iron

The maximum total iron concentration at SNP 02-15 (1,280 µg/L) occurred in January 2013 and was above the AO of 300 µg/L (Table 3-21). A second sample was collected in February 2013; total iron in that sample was lower (186 µg/L), and below the AO, likely due to the lower turbidity in the second sample. Turbidity in the first sample was 10 nephelometric turbidity units (NTU), compared to 1.3 NTU in the second sample and an average of <1 NTU in Snap Lake. Since the iron concentrations in both samples at SNP 02-15 were well above the maximum total iron concentration observed in Snap Lake (12 µg/L) in 2013, these samples were not representative of the water quality in Snap Lake. The elevated iron concentration in the first sample from SNP 02-15, of which 88% was in particulate form, was likely due to the higher turbidity. The filtration treatment at the potable water treatment plant is intended to reduce turbidity, and therefore particulate forms of metals, in drinking water. The elevated iron poses no health risk to humans, as the AO is based on taste and staining of laundry and plumbing fixtures (Health Canada 2012). However, water sampling procedures and the condition of the intake structure (i.e., screen, pump placement) will be investigated to assess the potential source of turbidity in the water sample.

Table 3-21: Comparison of Snap Lake Data to Canadian Drinking Water Quality Guidelines

Parameter	Units	Canadian Drinking Water[a]	Wildlife Health (Livestock)[b]	Type	Observed Data[c]					
					Snap Lake			SNP 02-15		
					Result	Percent of Drinking Water Guideline	Percent of Wildlife Guideline	Result	Percent of Drinking Water Guideline	Percent of Wildlife Guideline
Conventional Parameters										
Laboratory pH	unitless	6.5 to 8.5	-	range	6.8 to 7.8	-	-	6.9 to 7.5	-	-
Total Dissolved Solids, calculated (lab)	mg/L	≤500 (AO)	3,000	max	301	60%	10%	177	35%	6%
Major Ions										
Calcium	mg/L	-	1,000	max	61	-	6%	36	-	4%
Chloride	mg/L	≤250 (AO)	-	max	134	54%	-	77	31%	-
Fluoride	mg/L	1.5	2	max	0.23	15%	12%	0.13	9%	7%
Sodium	mg/L	≤200 (AO)	-	max	33	16%	-	20	10%	-
Sulphate	mg/L	≤500 (AO)	1,000	max	26	5%	3%	16	3%	2%
Nutrients										
Nitrate + nitrite	mg-N/L	-	100	max	3.06	-	3%	1.63	-	2%
Nitrate, as N, calculated	mg-N/L	10	-	max	3.04	30%	-	1.63	16%	-
Nitrite, as N	mg-N/L	1	10	max	0.027	3%	<1%	<0.05	<1%	<1%
Total Metals										
Aluminum	µg/L	-	5,000	max	8.00	-	<1%	5.47	-	<1%
Arsenic	µg/L	10	25	max	0.1	1%	<1%	0.1	1%	<1%
Antimony	µg/L	6	-	max	3.77	63%	-	0.063	1%	-
Barium	µg/L	1000	-	max	29	3%	-	18	2%	-
Beryllium	µg/L	-	100	max	<0.01	-	<1%	<0.01	-	-
Boron	µg/L	5000	5,000	max	69	1%	1%	36	<1%	<1%
Cadmium	µg/L	5	80	max	0.016	<1%	<1%	1.16	23%	1%
Chromium	µg/L	50[d]	50	max	0.18	<1%	<1%	0.24	<1%	<1%
Hexavalent Chromium	µg/L	50[d]	-	max	1.3	3%	-	-	-	-
Cobalt	µg/L	-	1,000	max	0.038	-	<1%	0.062	-	<1%
Copper	µg/L	≤1,000 (AO)	500	max	1.1	<1%	<1%	1.7	<1%	<1%
Iron	µg/L	≤300 (AO)	-	max	12	4%	-	*1,280*[e]	>100%	-
Lead	µg/L	10	100	max	0.03	<1%	<1%	4.14	41%	4%
Manganese	µg/L	≤50 (AO)	-	max	14	28%	-	10	20%	-
Mercury	µg/L	1	3	max	0.002	<1%	<1%	-	-	-
Molybdenum	µg/L	-	500	max	1.73	-	<1%	0.74	-	<1%
Nickel	µg/L	-	1,000	max	2.64	-	<1%	1.44	-	<1%
Selenium	µg/L	10	50	max	0.1	1%	<1%	<0.04	<1%	<1%
Uranium	µg/L	20	200	max	0.29	1%	<1%	0.11	<1%	<1%
Vanadium	µg/L	-	100	max	0.06	-	<1%	<0.05	-	-
Zinc	µg/L	≤5,000 (AO)	50,000	max	3	<1%	<1%	60	<1%	<1%

Table 3-21: Comparison of Snap Lake Data to Canadian Drinking Water Quality Guidelines

Parameter	Units	Canadian Drinking Water[a]	Wildlife Health (Livestock)[b]	Type	Observed Data[c]					
					Snap Lake			SNP 02-15		
					Result	Percent of Drinking Water Guideline	Percent of Wildlife Guideline	Result	Percent of Drinking Water Guideline	Percent of Wildlife Guideline
Organics-Volatiles										
Benzene	mg/L	0.005	-	max	<0.0005	10%	<1%	-	-	-
Ethylbenzene	mg/L	0.0024	2.4	max	<0.0005	21%	<1%	-	-	-
Toluene	mg/L	0.0024	24	max	0.00078	33%	<1%	-	-	-
Xylene	mg/L	0.3	-	max	0.0018	<1%	-	-	-	-
Microbiology										
E. coli	MPN/100 mL	0	-	max and range in whole-lake average[f]	*2* and <1 to <1	N/A	-	**1**[g]	N/A	-
Total coliforms	MPN/100 mL	0	-	max	-	N/A	-	**411**	N/A	-

Note: **Bold** values are above the relevant Maximum Acceptable Concentrations (MAC).

Italicized values are above the relevant Aesthetic objective (AO). Aesthetic effects (e.g., taste, odour) affect whether consumers will consider the water drinkable.

SNP 02-15 = water intake from Snap Lake.

a) Canadian drinking water quality guidelines (WQG) are obtained from Health Canada (2012). Unless stated, the WQG concentrations are MAC.

b) Wildlife health guidelines were based on livestock watering guidelines from CCME (1999).

c) Observed concentrations within the 2013 reporting period, which is November 1, 2012 to October 31, 2013 for Snap Lake and for SNP 02-15.

d) Although the WQG is protective of health effects from chromium (VI), it applies to total chromium including both chromium (III) and (VI).

e) The maximum total iron concentration was measured in January 2013 and was above the Canadian aesthetic objective, The elevated total iron concentration was not representative of water quality in Snap Lake and was likely due to the higher turbidity in the sample (10 NTU) relative to Snap Lake (average <1 NTU).

f) Minimum and maximum whole-lake average concentration calculated by area, excluding northwest arm stations.

g) The Whole-lake average calculation was not applicable to water intake station (SNP 02-15).

N/A = Action Levels do not apply for microbiological parameters; SNP = Surveillance Network Program; N = nitrogen; - = not applicable; < = less than; ≤ = less than or equal to; max = maximum; *E. coli* = *Escherichia coli*; µg/L = micrograms per litre; mg/L = milligrams per litre; MPN/100 mL = most probable number per 100 millilitres; NTU = nephelometric turbidity units.

3.4.7.1 Summary of Key Question 6

Concentrations of water quality parameters in Snap Lake were below drinking WQGs, with the exception of *E. coli* and total coliforms. Drinking water at the Mine is filtered and chlorinated prior to consumption, thus drinking water at the Snap Lake camp was acceptable from a microbiological perspective (i.e., *E. coli* and total coliforms).

Total iron in one sample at SNP 02-15 was above the AO; however, the elevated iron concentration was not representative of Snap Lake water quality and was likely due to high turbidity in the sample. Iron concentrations in a follow-up sample at SNP 02-15, with lower turbidity, indicated that iron was below the non-health-related AO. The filtration treatment reduces turbidity, and therefore particulate forms of metals in the drinking water. The camp workers who drink the water are not at risk and Snap Lake water is safe for humans (pending disinfection). Drinking water at the Mine will continue to be tested regularly and results reported to the local Health Authority.

3.4.8 Action Level Summary

Low Action Levels related to toxicological impairment that could affect ecological function in Snap Lake were triggered for three parameters: chloride, nitrate, and fluoride:

- Maximum monthly concentrations were above 75% of generic AEMP benchmarks (see Section 3.4.3).
- Concentrations were increasing over time in Snap Lake (see Section 3.4.4).
- Concentrations were greater in Snap Lake relative to both reference lakes (see Section 3.4.5).

Results of toxicity testing did not trigger Action Levels because toxicity testing did not show any toxic effects to test organisms in the mixing zone samples (see Section 3.4.3).

No Action Levels were triggered for nutrient enrichment because maximum 2013 whole-lake average concentrations of phosphorus and nitrogen were within EAR predictions (see Section 3.4.4) and the 2013 maximum whole-lake average concentration of TP did not exceed 75% of the AEMP nutrient benchmark for TP (see Section 3.4.3).

De Beers will investigate the cause of the high iron concentration observed at SNP 02-15, through a review of water sampling procedures and the intake structure (i.e., screen condition, pump location) to determine whether adjustments or maintenance are appropriate. Filtration provided by the potable WTP and the lack of human health concerns with iron concentrations above the AO supports the conclusion that the Mine’s treated drinking water poses no risk to human health. As such, no further action is required as water from Snap Lake is considered safe to drink.

3.5 Conclusions

3.5.1 Key Question 1: Are Concentrations or Loads of Key Water Quality Parameters in Discharges to Snap Lake Consistent with EAR Predictions and Below Water Licence Limits?

Clear increasing or decreasing trends in concentrations were not observed for many signature parameters; however, loadings to Snap Lake have increased due to increases in daily discharge rates. In 2013, the annual treated effluent volume was approximately 31% higher than in the 2012.

Chemical signatures of treated effluent from the Mine are TDS and its component ions (calcium, chloride, fluoride, magnesium, nitrate and nitrite, potassium, sodium, and sulphate), nitrogen nutrients (ammonia, nitrate, and nitrite), and nine metals (barium, boron, lithium, manganese, molybdenum, nickel, rubidium, strontium, and uranium). Concentrations in the treated effluent remained below the maximum allowable concentration in any grab sample of treated effluent for all parameters in 2013 with the exception of chloride. Chloride concentrations were above the AML in September 2013, but below the AML from October to December 2013.

The 2013 flow-weighted average concentration of sulphate was above the maximum average annual concentration predicted in the EAR; sulphate concentrations have historically been above EAR predictions. The CCME currently does not provide WQGs for sulphate. However, sulphate is a component of TDS (i.e., approximately 9%), so it was implicitly considered as part of the aquatic toxicity testing conducted to develop an appropriate site-specific, effects-based TDS water quality benchmark. The flow-weighted average concentrations were below or slightly above the upper bound of predicted average concentrations used in the 2013 modelling update, with the exception of aluminum, chromium, iron, and lead. Uncertainties related to groundwater inflows may have resulted in the under-prediction of these metals in the 2013 model.

The 2013 TP, nitrate, and ammonia annual loadings to Snap Lake from the WTP were well below the Water Licence limits.

The 2013 treated effluent samples did not show any acute toxicity response for either Rainbow Trout or *Daphnia magna*. The regulatory requirement to demonstrate an absence of acute toxicity to juvenile Rainbow Trout (MVLWB 2004, 2012) was confirmed. Acute toxicity has not occurred in any of the treated effluent samples collected from 2005 to 2013.

Chronic toxicity was predicted to occur in treated effluent in the EAR (De Beers 2002). In 2013, a treated effluent sample from the permanent WTP showed no evidence of chronic toxicity to *Ceriodaphnia dubia* or algal growth inhibition. Most of the algal tests performed on treated effluent showed growth stimulation.

3.5.2 Key Question 2: Are Concentrations of Key Water Quality Parameters in Snap Lake below AEMP Benchmarks and Water Licence Limits?

The 2013 water quality data from Snap Lake were below AEMP benchmarks and Water Licence limits, with the exception of chloride, fluoride, and nitrate. Snap Lake concentrations for chloride, fluoride and nitrate were above the generic CCME WQGs used for AEMP benchmarks, but below the recommended SSWQOs. Since the primary source of fluoride, chloride, and nitrate was treated effluent, increases are associated with elevated calcium and hardness, which are expected to reduce the potential for toxicity effects associated with fluoride, chloride, and nitrate.

Low field pH values were measured in Snap Lake and the reference lake, Northeast Lake, in 2013, and were occasionally below the CCME WQG of 6.5. Low field pH values have been also measured in Snap Lake in previous years during Mine operations and in baseline conditions during the ice-covered season. Both field pH and laboratory pH values have increased over time in Snap Lake to above the normal and the reference lakes range. The occasional low pH values in 2013 were likely due to natural variation in pH values because the overall trend in pH in Snap Lake is increasing.

In 2013, DO concentrations in Snap Lake were considered healthy for fish and other aquatic organisms, with the exception of six locations, where field DO readings dropped below the CCME WQG of 6.5 mg/L. At half of these locations, the low DO was limited to the bottom 0.5 m of the water column, indicating that the probe was likely near the sediment boundary, or submerged in sediment. The other three locations where DO was below the CCME WQG were the deepest stations in Snap Lake; the low readings occurred during ice-covered conditions. Concentrations of DO below the CCME WQG were observed near bottom during ice-covered conditions in both Snap Lake during baseline conditions and the reference lakes in 2013. Overall, DO concentrations in Snap Lake do not appear to have decreased as a result of treated effluent discharge. In 2013, increases rather than reductions in bottom DO concentrations were observed around the diffuser relative to the northwest arm.

Whole-lake average TDS concentrations ranged from 228 mg/L to 284 mg/L, with a maximum TDS concentration of 301 mg/L, all below the Water Licence limit of 350 mg/L, and the recommended SSWQO (i.e., 684 mg/L).

Toxicity test results from three diffuser samples collected in May and three collected in September showed no adverse effects for any test endpoints. Algal growth was stimulated in all samples; the degree of stimulation increased at higher sample concentrations (Appendix 3F).

3.5.3 Key Question 3: Which Water Quality Parameters are Increasing over Time in Snap Lake, and How do Concentrations of these Parameters Compare to AEMP Benchmarks, Concentrations in Reference Lakes, EAR Predictions, and Subsequent Modelling Predictions?

The EAR predicted increases in concentrations of major ions, nutrients, and metals in Snap Lake due to the discharge of treated effluent over time (De Beers 2002). The EAR also predicted that DO would decrease over time in Snap Lake as a result of the treated effluent discharge. In 2013 the following parameters increased above the Snap Lake normal range (i.e., baseline mean ± two SDs) and reference lakes (Northeast Lake and Lake 13) concentrations in at least one area of Snap Lake:

- total alkalinity, TDS, reactive silica, and total hardness;
- eight major ions (bicarbonate, calcium, chloride, fluoride, magnesium, potassium, sodium, and sulphate);
- all monitored nitrogen parameters (TKN, ammonia, nitrate, and nitrite); and,
- eight metals (barium, boron, lithium, molybdenum, nickel, rubidium, strontium, and uranium).

Other parameters that increased in Snap Lake, but were not strongly correlated with conductivity, were laboratory and field pH, TKN and total manganese. Phosphorus concentrations have not increased in Snap Lake.

Whole-lake averages in 2013 were slightly above updated 2013 model predictions for barium and uranium, likely due to model uncertainties. The 2013 whole-lake average for antimony was well above the updated 2013 model prediction value. Since antimony concentrations have not increased in Snap Lake and were similar to reference lakes, the difference in observed and predicted values was likely related to a contamination issue identified in the QA/QC analysis.

The 2013 whole-lake averages for parameters that were above AEMP benchmarks (i.e., chloride, fluoride, and nitrate) were below the available EAR predictions and the upper range of the 2013 predictions. Concentrations of TDS, a key indicator of Mine-related changes to water quality in Snap Lake, were also below the EAR predictions and the upper range of the 2013 predictions.

Minimum field pH values, which were below the AEMP benchmark, could not be compared to model predictions because pH was not modelled in the EAR or 2013 model. However, the observed increasing trends in both field and laboratory pH values indicates that the low pH values are not due to a decreasing trend in pH.

In 2013, increases in surface and bottom water DO concentrations were measured during ice-covered conditions in the main basin of Snap Lake. The increase in bottom DO concentrations during ice-covered conditions near the diffuser may result from the release of oxygenated treated effluent from the diffuser near the lake bottom.

3.5.4 Key Question 4: Are Spatial and Seasonal Patterns in Water Quality in Snap Lake and Downstream Waterbodies Consistent with Predictions presented in the EAR and Subsequent Modelling Predictions?

Spatial and seasonal patterns were observed for some water quality parameters in Snap Lake. The patterns observed in 2013 can be explained by the discharge of treated effluent, seasonal differences in mixing conditions in Snap Lake, and natural biological and physical processes within the lake.

Horizontal patterns involved gradual declines in concentration with increasing distance away from the diffuser for TDS and a number of other water quality parameters directly associated with treated effluent discharge (conductivity, most major ions, nitrogen [nitrate, nitrite, ammonia], and eight metals that correlated with conductivity [barium, boron, lithium, molybdenum, nickel, rubidium, strontium, and uranium]). Concentration gradients within the main basin of Snap Lake for these parameters were less prominent in 2013 compared to gradients observed in the first four years of minewater discharges to Snap Lake (i.e., 2004 and 2007).

Concentrations of most treated effluent related parameters in the northwest arm continue to be notably lower compared to the main basin due to the limited hydraulic connection between the northwest arm and the main basin. However, the lower concentrations observed in the northwest arm are now higher than those observed for Northeast Lake and Lake 13. Higher concentrations closer to the northwest arm's narrow connection to the main basin were evident again in 2013.

Manganese and cobalt were higher in Snap Lake relative to the reference lakes in 2013. Although these two metals are at similar concentrations in the effluent as barium and uranium, respectively, manganese and cobalt were not positively correlated with conductivity. Correlation with conductivity has typically been used as an indicator of treated effluent; however, these two metals may be behaving less conservatively than other effluent-related parameters due to oxidation and reduction processes, controlled partly by DO concentrations, in Snap Lake.

The lack of a clear spatial patterns in phosphorus was consistent with previous spatial assessments, and likely indicates that either phosphorus removal processes are reducing phosphorus concentrations in the water column (i.e., through phytoplankton uptake and/or sedimentation processes) or differences are too subtle to observe, given the uncertainty in low level phosphorus measurements.

Vertical patterns in field conductivity in 2013 indicated that the plume may no longer be sinking to the bottom of Snap Lake due to a lower density difference between the plume and lake water. Open-water profiles of conductivity indicate that the plume continues to be more evenly mixed throughout the water column during open-water conditions, with the exception of deeper locations. These deeper locations appear to be less influenced from wind-driven mixing; the presence of naturally occurring thermoclines at these locations may be further inhibiting mixing. Vertical gradients in DO and water temperature were observed, typically during ice-covered conditions, related primarily to natural lake processes. Locations close to the discharge appear to have higher DO concentrations due to the discharge of well-oxygenated effluent.

Seasonal differences between ice-covered and open-water conditions were less prominent in 2013 compared to 2004 to 2007. The reduction in the range of seasonal and spatial differences is attributed to the greater mixing of the treated effluent discharge in Snap Lake.

Results from the Downstream Lakes Special Study (Section 11.3) showed evidence of the treated effluent throughout lakes DSL1 and DSL2, and near the inlet of Lac Capot Blanc in 2013. Concentrations of Mine-related constituents reached background concentrations approximately 11 km downstream of Snap Lake, which is 5 km further downstream than in 2013. In the EAR, parameter concentrations associated with the treated effluent discharge were conservatively predicted to reach near background concentrations within approximately 44 km of Snap Lake, assuming maximum concentrations during operations. An increasing trend in TDS was observed at King Lake, which is 25 km downstream; however, 2013 concentrations of TDS in King Lake remained within background levels.

3.5.5 Key Question 5: Is there Evidence of Acidification Effects from the Mine on Nearby Waterbodies?

Based on the 2013 data, there was no strong evidence of acidification of inland lakes IL3, IL4, and IL5. These results are consistent with previous assessments, which concluded that there is limited potential for acidification of these lakes due to emissions from the Mine. Water quality in these lakes may be changing over time, most noticeably at IL5, where concentrations of sulphate, base captions, and alkalinity are elevated compared to baseline. Laboratory pH values have been decreasing in all three lakes; however, field pH values were variable in the inland lakes. The differences between laboratory and field pH values will be investigated in 2014 (Appendix 3A) to assess the differences in observed trends.

The ranges in pH, total alkalinity, sulphate, and base cations in Streams S1 and S27 in 2013 were similar to baseline values; therefore, no evidence of acidification, or any spring acid pulse, was discernible in Streams S1 and S27 in 2013.

3.5.6 Key Question 6: Is Water from Snap Lake Safe to Drink?

Concentrations of most water quality parameters in Snap Lake were below health-based drinking water guidelines, with the exception of *E. coli* and total coliforms. Drinking water at the Mine is filtered and chlorinated prior to consumption, thus drinking water at the Snap Lake camp was acceptable from a microbiological perspective (i.e., *E. coli* and total coliforms). Based on the information available, Snap Lake water is considered safe for humans (pending disinfection) and wildlife to drink.

3.5.7 Action Level Assessment

Low Action Levels related to toxicological impairment in Snap Lake were triggered for three parameters: chloride, nitrate, and fluoride. Response plans for chloride, nitrate, and fluoride are discussed in Section 13. No Action Levels related to nutrient enrichment or toxicity were triggered.

While an elevated iron concentration was above the AO at the raw water intake station (SNP 02-15), it was likely due to high turbidity in the sample; the source of the turbidity will be further investigated. Filtration provided by the potable water treatment plant and the lack of human health concerns with iron concentrations above the AO support the conclusion that the Mine's treated drinking water poses no risk to human health. As such, no further action is required as water from Snap Lake is considered safe to drink pending disinfection.

3.6 Recommendations

The following are recommended for the water quality component of the 2013 AEMP:

Data Quality and Continual Improvement

- Implement the recommendations from the QA/QC assessment (outlined in Appendix 3A), which focuses on investigating potential contamination and variability between samples. These recommendations include reducing variability between field DO and Winkler titration DO, and continuing to consider options for minimizing holding time issues for parameters with sensitive holding times (e.g. freezing samples), and discussing analytical procedures with the laboratories, particularly for antimony, to determine potential sources and/or interferences that may be contributing to measured blank concentrations.
- Continue to periodically investigate the accuracy and precision of analyzing TP by the analytical laboratories currently used in the AEMP program and the potential for streamlining the collection of nutrient data by the water quality and plankton components (outlined in Appendix 3B). A limited number of nutrient spike samples should routinely be sent to the primary laboratories used for nutrient analyses in the AEMP as an on-going and independent check of the accuracy of nutrient results. Recommendations are completing one season of split sampling at plankton stations, and sending split samples to the two primary laboratories that provide nutrient analyses for the water quality and plankton sections. The split samples are intended to provide the plankton component with sufficient overlapping data to merge historical plankton nutrient data analyzed by UofA with future plankton data recommended to be analyzed by ALS. The splits samples will also be used to confirm whether the lack of differences between mid-depth and depth-integrated samples for nutrients is applicable in Northeast Lake.
- Identify the potential cause(s) of high turbidity at SNP 02-15 by assessing whether sampling procedures or the location or condition of the water intake structure may be introducing turbidity in samples collect at SNP 02-15. Based on those findings, review whether data from SNP 02-15 are appropriate to determine whether water in Snap Lake is safe to drink.

Water Quality Data Interpretation

- Give consideration to parameters with concentrations that have increased beyond the normal range in Snap Lake, but for which there are either no relevant AEMP benchmarks (i.e., barium, lithium, rubidium), or the recommended site-specific benchmark has not yet been accepted (i.e., strontium). It is recommended that available toxicological literature be reviewed to determine the implications of increases in total barium, lithium, and rubidium on aquatic life.

Water Quality Prediction Refinement

- Continue to make necessary adjustments to loadings and predictions for TDS and other treated effluent-related parameters. The re-evaluation of the predicted loadings and consequences to the water quality in Snap Lake are being conducted because the concentrations of TDS and other treated effluent-related parameters are directly correlated to increased loadings.

3.7 References

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SECTION 4

SEDIMENT QUALITY

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List of Acronyms

AEMP	Aquatic Effects Monitoring Program
ALS	ALS Canada Ltd.
CCME	Canadian Council of Ministers of the Environment
CCMS	collision cell inductively coupled plasma mass spectrometry
D	diffuser
De Beers	De Beers Canada Inc.
DL	detection limit
DO	dissolved oxygen
DQO	data quality objective
dw	dry weight
EAR	Environmental Assessment Report
e.g.	for example
Golder	Golder Associates Ltd.
GPS	global positioning system
ICP-MS	inductively coupled plasma mass spectrometry
ISQG	Interim Sediment Quality Guideline
Max	maximum
MB	Main Basin of Snap Lake
Min	minimum
Mine	Snap Lake Mine
MVLWB	Mackenzie Valley Land and Water Board
LK13	Lake 13
N	nitrogen
n	sample size
NWA	Northwest Arm
P	phosphorus
P	probability
PAH	polycyclic aromatic hydrocarbon
PEL	Probable Effect Level
QA/QC	quality assurance and quality control
RPD	relative percent difference
S	sulphur
SD	standard deviation
SNAP	Snap Lake
SNP	Surveillance Network Program
SQG	sediment quality guideline
TKN	total Kjeldahl nitrogen
TOC	total organic carbon
UTM	Universal Transverse Mercator

Units of Measure

≤	less than or equal to
<	less than
>	greater than
±	plus or minus
%	percent
% dw	percent dry weight
°C	degrees Celsius
µS/cm	microSiemens per centimetre
cm	centimetre
m	metre
mm	millimetre
m^2	square metres
mg/kg	milligrams per kilogram
mg/kg dw	milligrams per kilogram dry weight

4 SEDIMENT QUALITY

4.1 Introduction

4.1.1 Background

This section presents sediment quality data collected from Snap Lake and two reference lakes (Northeast Lake and Lake 13) in 2013. This section provides results of comparisons to Canadian sediment quality guidelines (SQGs) for protection of freshwater aquatic life (CCME 1999 with updates), and the results of analyses of patterns in concentrations of target sediment quality parameters. Temporal trends in sediment concentrations at the Snap Lake diffuser were evaluated and are reported herein.

Sediment quality has been monitored as part of the Aquatic Effects Monitoring Program (AEMP) since 2004. A brief overview of the history and development of the sediment quality sampling program is provided below.

Snap Lake sediments were sampled in 1999 to assess baseline sediment quality and complete the Environmental Assessment Report (EAR) for the Snap Lake Mine (Mine). Those samples were collected at three relatively shallow stations located in near-shore areas and at one station in the main basin area (De Beers 2002). Sediments were also sampled in 2004 before discharge of treated effluent to Snap Lake began, providing an additional year of baseline sediment quality data. The 2004 baseline sediment sampling was conducted at stations representative of deeper water locations than those currently being monitored.

The first sediment quality monitoring event to occur after discharge of treated effluent to Snap Lake began in April 2005. At that time, sediment quality monitoring focused on locations with water depths that ranged from 4 to 22 metres (m). In 2006 and 2007, benthic invertebrate sampling focused on stations in water depths between 10 and 15 m; sediments were also sampled at these locations. Sediment quality monitoring stations with depths less than 10 m were replaced by locations with water depths between 10 and 15 m to eliminate the potentially confounding effect of varying water depth on benthic invertebrate community structure. The sediment sampling program was limited to stations within Snap Lake until 2007, but was expanded in 2008 to include Northeast Lake as a reference lake. The sediment sampling program was further expanded in 2012 to include Lake 13 as a second reference lake.

Prior to 2007, sediment sampling involved collection and processing of entire Ekman grabs of sediment, which were referred to as bulk sediment samples. Because sedimentation rates in arctic and sub-arctic lakes tend to be very slow, the 10 to 15 centimetres (cm) depth of sediment typically retrieved by an entire Ekman grab can represent several decades of sediment deposition (MacDonald 1983; Wolfe et al. 1996; Vardy et al. 1997; Szeicz and MacDonald 2001). Therefore, bulk sediment quality data collected before 2007 were likely dominated by sediment chemistry more characteristic of the period before Mine activity. Sediment collection procedures were modified in 2007 and 2008 to target more recent changes in sediment chemistry, such that only the top 5-cm of sediment was retained from each Ekman grab. To allow a comparison of sediment chemistry between the two sampling techniques in 2007 and 2008, all stations were sampled for the top 5-cm of sediment, and bulk samples were also collected at a subset of nine stations. Results of this comparison of the two sampling techniques indicated that mean parameter concentrations were generally similar and that discontinuing the bulk sampling should provide more representative data without adversely affecting comparability to previous years' sediment chemistry data (De Beers 2009). Therefore, each station has been sampled for the top 5-cm of sediment since 2009.

The sediment quality monitoring program was altered in 2009 with respect to the timing of sample collection (De Beers 2010). Prior to 2009, this work was conducted in March/April under late winter conditions while ice cover was still present. Treated effluent concentrations were expected to be highest and dissolved oxygen (DO) concentrations lowest in late winter, and therefore likely to have the greatest potential for effects on the benthic invertebrate community. Following the 2008 monitoring program it was decided that the benthic invertebrate sampling program should be moved to late summer for 2009 and subsequent years. Logistical issues associated with winter field work prevented completion of the benthic sampling program in some years and the anticipated low DO concentrations were not observed during winter. Therefore the original reason for conducting the benthic program during winter conditions was no longer applicable. The sediment sampling program, which is conducted in conjunction with benthic invertebrate sampling, was moved to late summer as well.

In 2011 and 2012, separate sampling trials were undertaken at a total of six stations to determine whether the depth of sediment sampled could be further reduced and whether this would reflect a difference in sediment chemistry results. Sediments were collected using an Ekman grab to sample the top 5-cm of sediment and a Tech-Ops sediment corer to sample the top 2-cm of sediment. The top 2-cm layer was the thinnest layer that could be sampled reliably, because of the soft, unconsolidated nature of the sediments in Snap Lake. The Snap Lake diffuser station, SNP 02-20e, was the only one to show a depth-related difference with higher concentrations of a majority of parameters occurring in the shallower 2-cm layer.

The 2013 AEMP was the first year of sediment quality monitoring under the new AEMP Design Plan (De Beers 2014), which called for monitoring the Snap Lake diffuser station (SNP 02-20e) at two sediment depths annually, and monitoring all the AEMP stations every three years; the next full sediment quality monitoring event will be in 2015. The planned sediment sampling program for 2013 was expanded to address other data needs. Lake 13 stations were sampled in 2013 to assess anomalous results for several parameters reported in 2012 (De Beers 2013), and to provide data to support the winter road study; other stations in Snap Lake and Northeast Lake were sampled to support the 2013 AEMP benthic invertebrate component.

4.1.2 Objectives

The overall objective of the AEMP sediment quality monitoring program is to determine whether sediment quality in Snap Lake remains acceptable such that a healthy benthic invertebrate community is maintained. The specific objectives of the re-designed sediment quality monitoring program were:

- to characterize and interpret bottom sediment quality in Snap Lake and two reference lakes, and make comparisons to previous years;
- to verify predictions made in the EAR (De Beers 2002) about Mine effects on lake bottom sediment quality; and,
- to recommend any necessary changes to the sediment quality component of the AEMP for future years.

The Snap Lake sediment quality monitoring component of the AEMP was designed to meet the conditions of Part G of the Water Licence (MVLWB 2013).

Analysis of the sediment quality data is intended to address the following key questions:

- Are concentrations of sediment quality parameters above or below SQGs?
- Are there differences in sediment quality in Snap Lake relative to the reference lakes and, if so, are they related to the Mine?
- Are concentrations of sediment quality parameters increasing over time?

4.2 Methods

4.2.1 Field Survey

4.2.1.1 Sampling Locations

Sediments were sampled annually in five areas of Snap Lake from 2005 to 2012: the diffuser mixing zone, which is referred to as “diffuser” hereafter; the near-field, mid-field, and far-field areas of the main basin; and, the northwest arm. The northwest arm of Snap Lake was considered a reference area for the 2005 and 2006 sediment quality programs, but can no longer be considered a reference area as it has been exposed to treated effluent since 2007.

Sampling designs of the sediment quality and benthic invertebrate components of the AEMP changed from 2005 to 2012 to reflect temporal variation in exposure of the lake bottom to treated effluent. Baseline sediment sampling was performed at 12 Snap Lake stations in 2004. Modifications were made to the sampling design in 2005 and 2006, increasing the number of stations from 12 to 18 to provide broader spatial coverage of Snap Lake, and adjusting water depth of some locations for benthic invertebrate sampling. Two station locations were adjusted in 2007 to accommodate the May 2006 commissioning of the temporary diffuser at a slightly different location from the permanent diffuser and the SNAP15 station could not be sampled in 2007 due to unsafe conditions, but otherwise the same sediment quality stations were monitored in Snap Lake from 2007 to 2012.

Under the 2013 AEMP Design Plan (De Beers 2014), the diffuser and northwest arm stations remained unchanged, but the near-field, mid-field, and far-field areas were grouped together as one main basin area, and the number of sediment quality stations in the main basin was reduced from 14 to 6. The main basin stations retained for sediment quality monitoring were SNAP03, SNAP05, SNAP06, SNAP08, SNAP09, and SNAP11A. The Snap Lake stations sampled between 2004 and 2013 are shown in Table 4-1, and locations of the stations sampled in 2013 are shown in Figure 4-1.

Reference lake sampling has been part of the AEMP sediment quality component since 2008. Five stations have been sampled in Northeast Lake since 2008 (Figure 4-2), and five stations were sampled in Lake 13 in 2012 (Figure 4-3) to evaluate its suitability as a second reference lake.

Sediments were sampled for the following purposes in 2013:

- Sediment Depth Comparison: Sediment samples collected from the Snap Lake diffuser station, SNP 02-20e, to compare sediment quality in samples collected from the top 5-cm and top 2-cm layers of sediment. These samples were analyzed for the standard suite of AEMP sediment quality parameters: moisture content; particle size; total organic carbon (TOC); nutrients; and, metals[1].
- Lake 13 Reference Lake Sampling: Sediment samples collected from five stations in Lake 13 to assess the suitability of Lake 13 as a second reference lake for the AEMP and to provide data to support the winter road study. These samples were analyzed for the standard suite of AEMP sediment quality parameters, and also analyzed for polycyclic aromatic hydrocarbons (PAHs) to provide data for the winter road study.
- Sediment Sampling to Support Benthic Invertebrate Component: Sediment samples collected from ten stations in the northwest arm and main basin areas of Snap Lake and five stations in Northeast Lake. Because these stations were only sampled to support the benthic invertebrate component, benthic stations SNAP07 and SNAP15 were sampled and sediment quality station SNAP08 was not. These samples were only analyzed for moisture content, particle size, and TOC.

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[1] The list of elements reported in the total metals analysis includes metalloids such as arsenic and non-metals such as selenium, which are collectively referred to as "metals" in this chapter.

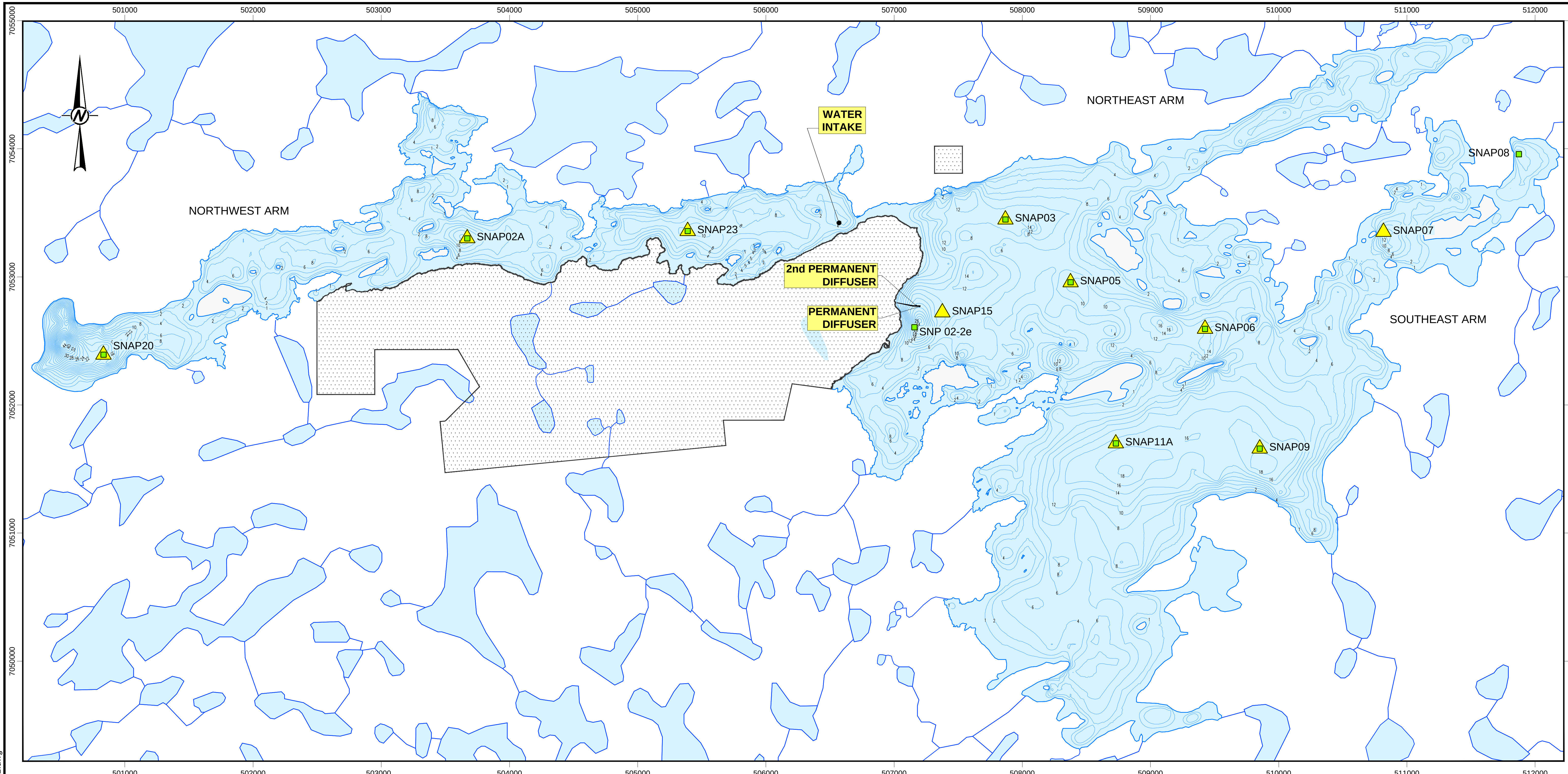
Table 4-1 Sediment Quality Stations Sampled in Snap Lake Since 2004

Lake Area [a]		Station	2004	2005	2006	2007	2008 to 2012	2013
Diffuser		SNP 02-20a	–	X	–	–	–	–
		SNP 02-20b	–	X	X	–	–	–
		SNP 02-20e	–	–	–	X	X	X
Main Basin	Near-field	SNAP03	X	X	X	X	X	X
		SNAP05	X	X	X	X	X	X
		SNAP06	X	X	X	X	X	X
		SNAP12	X	X	X	X	X	–
		SNAP13	X	X	X	–	–	–
		SNAP14	X	–	X	X	X	–
		SNAP14A	–	X	–	–	–	–
		SNAP15	–	–	X	–	X	X [b]
		SNAP16	–	X	–	–	–	–
		SNAP26	–	–	–	X	X	–
	Mid-field	SNAP09	X	X	X	X	X	X
		SNAP11 / 11A	X	X	X	X	X	X
		SNAP17	–	–	X	X	X	–
		SNAP18	–	–	X	X	X	–
		SNAP19	–	–	X	X	X	–
	Far-field	SNAP04	–	X	–	–	–	–
		SNAP07	X	X	X	X	X	X [b]
		SNAP08	X	X	X	X	X	– [b]
		SNAP10	–	X	–	–	–	–
Northwest Arm		SNAP01	X	X	–	–	–	–
		SNAP02 / 02A	X	X	X	X	X	X
		SNAP20	–	–	X	X	X	X
		SNAP23	–	–	X	X	X	X

a) The former near-field, mid-field, and far-field areas were grouped as the main basin in 2013, and the number of stations was reduced.

b) Main basin stations were only sampled to support the benthic invertebrate component in 2013; therefore, benthic stations SNAP07 and SNAP15 were sampled and sediment quality station SNAP08 was not.

X = station was sampled; – = station was not sampled.



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LEGEND

- Depth Contour (m)
- Watercourse
- Waterbody
- Snap Lake Mine Footprint
- Sediment Quality Monitoring Station
- Benthic Invertebrate Monitoring Station

REFERENCES

Digital Map from Mackay lake, Northwest Territories, produced by Department of Energy, Mines and Resources. Map 75M, Original scale 1:250,000, projection : Transverse Mercator, Datum : NAD83, coordinate system : UTM zone 12.

Snap Lake outline and islands were corrected to LANDSAT 7 satellite image 45/15, dated September 2, 2000. provided by Geobase.

Bathymetry was created in Surfer 8 using sonar data from the 2002 North Lakes program (Golder) and 2005 transect data from the Reference Lake Search program (Golder).

NOTES

Map moved to projection : Transverse Mercator, Datum : NAD83,
Coordinate system : UTM zone 12.

SNAP LAKE MINE

Sediment Quality Monitoring Stations in Snap Lake, 2013

PROJECTION: UTM Zone 12	DATUM: NAD83
SCALE 1:30,000 (500 0 500 METRES)	DE BEERS GROUP OF COMPANIES
FILE No: 1313490001-7010_4-1	DATE: February 24, 2014
JOB No: 14-1349-0004	REVISION No: A
OFFICE: GOLDER VICTORIA	DRAWN: JEF
CHECK: PMC	Figure 4-1

LEGEND

- Depth Contour (m)
- Watercourse
- Waterbody
- Sediment Quality Monitoring Station

REFERENCES

Digital Map from Mackay lake, Northwest Territories, produced by Department of Energy, Mines and Resources. Map 75M, Original scale 1:250,000, projection : Transverse Mercator, Datum : NAD83, coordinate system : UTM zone 12.

Lake outline and islands were corrected to LANDSAT 7 satellite image 45/15, dated September 2, 2000. provided by Geobase.

Bathymetry was created in Surfer 8 using sonar data from the 2002 North Lakes program (Golder) and 2005 transect data from the Reference Lake Search program (Golder).

NOTES

Grid is displayed in Transverse Mercator, Datum : NAD83, Coordinate system : UTM zone 12.

SNAP LAKE MINE

Sediment Quality Monitoring Stations in Northeast Lake, 2013

PROJECTION: UTM Zone 12	DATUM: NAD83	DE BEERS GROUP OF COMPANIES	
SCALE 500 0 500 METRES			
FILE No: 1313490001-7010_4-2		DATE: February 24, 2014	
JOB No: 14-1349-0004	REVISION No: A	Figure 4-2	
OFFICE: GOLDER VICTORIA	DRAWN: JEF	CHECK: PMC	

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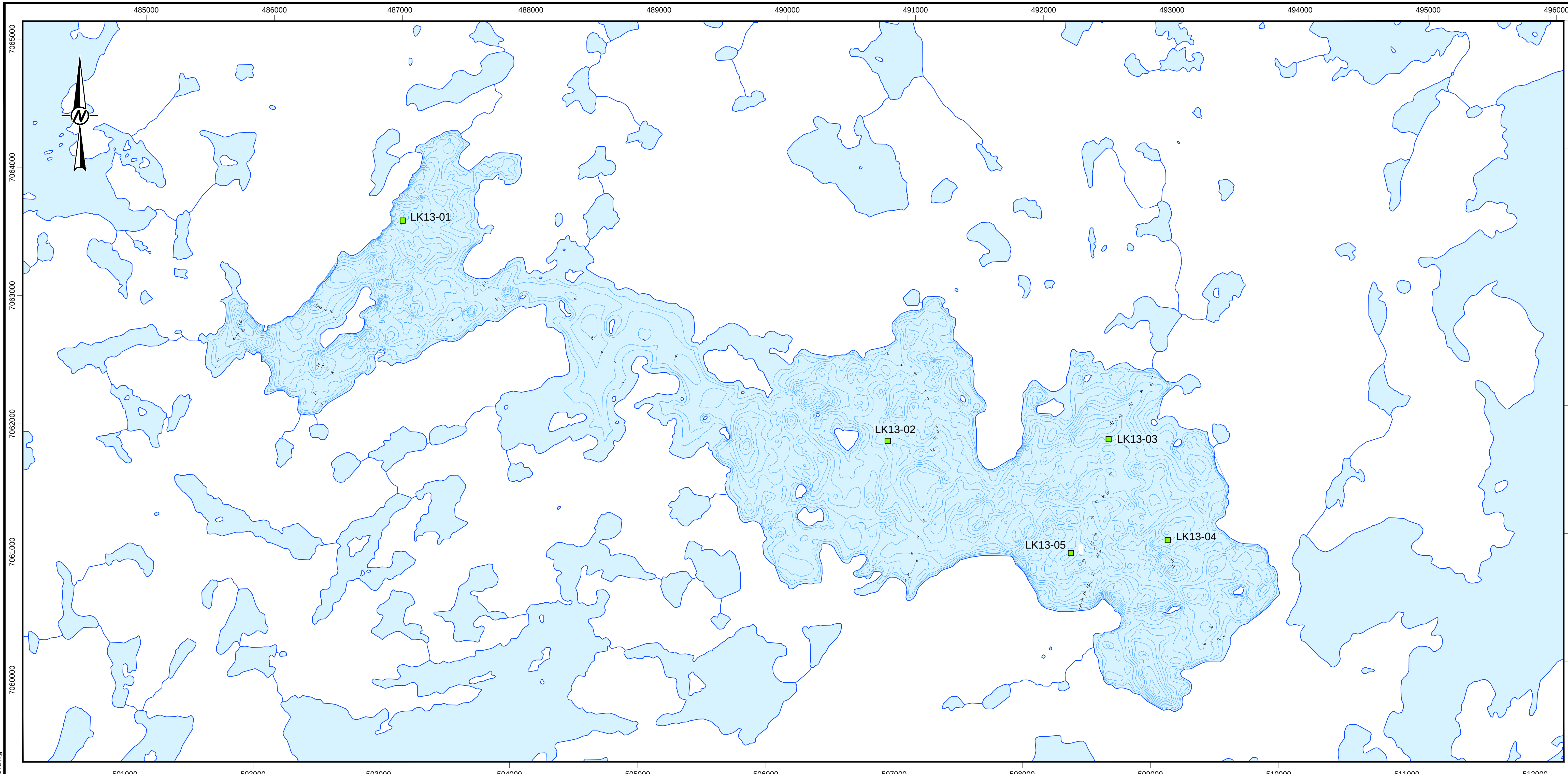

LEGEND

- Depth Contour (m)
- Watercourse
- Waterbody
- Sediment Quality Monitoring Station

REFERENCES

Digital Map from Mackay lake, Northwest Territories, produced by Department of Energy, Mines and Resources. Map 75M, Original scale 1:250,000, projection : Transverse Mercator, Datum : NAD83, coordinate system : UTM zone 12.

Bathymetry was created in Surfer 8 using sonar data from the 2002 North Lakes program (Golder) and 2005 transect data from the Reference Lake Search program (Golder).

NOTES

Map moved to projection : Transverse Mercator, Datum : NAD83, Coordinate system : UTM zone 12.

SNAP LAKE MINE

Sediment Quality Monitoring Stations in Lake 13, 2013

PROJECTION: UTM Zone 12		DATUM: NAD83	DE BEERS GROUP OF COMPANIES
500 0 500 SCALE METRES			
FILE No: 1313490001-7010_4-3			DATE: Februray 24, 2014
JOB No: 14-1349-0004	REVISION No: A		Figure 4-3
OFFICE: GOLDER VICTORIA	DRAWN: JEF	CHECK: PMC	

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4.2.1.2 Timing of Sampling

Sediment samples were collected during late summer, when ice-cover was absent on the lakes and treated effluent was discharging through the permanent diffuser. Sediment samples were collected between September 5 and 15, 2013.

4.2.1.3 Sampling Methods

Sediment sampling stations in Snap Lake were accessed by boat. A helicopter was used to transport the boat and field crew to Northeast Lake and to Lake 13.

At each station, three sediment grabs were collected using a 15-cm Ekman grab that samples an area of 0.023 square metres (m^2). The grab was thoroughly rinsed with lake water before sampling. After a sediment grab sample was collected, as much overlying water as possible was drained off without disturbing the sediment surface. If the surface of the retrieved sediment sample was disturbed, either during the initial sample collection or during the draining of overlying water, the sample was discarded and another grab sample was collected. At each station, the top 5-cm of sediment was removed from each of the three grabs using a clean stainless steel spoon and placed into a clean plastic container. Once this portion of sediment had been removed from all three grabs, the sediments were mixed until homogeneous in colour and texture to generate one composite sediment sample for each station, and then transferred to sample containers for delivery to the analytical laboratory.

For the sediment depth comparison at SNP 02-20e, three Ekman grabs were collected and processed as described above to generate a top 5-cm composite sediment sample. To sample the top 2-cm layer of sediment at the station, a 10-cm diameter Tech-Ops sediment corer was used. Three core samples were collected at the station; sediments were extruded from the core tube and the top 2-cm layer of sediment from all three cores was removed, homogenized to generate a composite sediment sample, and transferred to sample containers for delivery to the analytical laboratory.

Field duplicate samples were collected at two stations, using separately collected sets of three Ekman grab samples to sample the top 5-cm of sediment. The field duplicate stations for 2013 were SNP 02-20e and LK13-01.

4.2.1.4 Laboratory Analyses

Composite sediment samples were stored at 4 degrees Celsius (°C) and shipped on ice to the ALS Canada Ltd. (ALS) analytical laboratory in Edmonton, Alberta, for analyses of particle size, nutrients, carbon content, total metals, and/or PAHs, depending on where the samples were collected. The full parameter list is provided in Table 4-2.

Table 4-2 2013 Sediment Quality Parameter List for Samples Collected in Snap Lake, Northeast Lake, and Lake 13

Parameter Group	Parameter [Units]
Physical	moisture (%)
	gravel (% dw)
	sand (% dw)
	silt (% dw)
	clay (% dw)
Carbon	inorganic carbon (% dw)
	total carbon (% dw)
	total organic carbon (% dw)
Nutrients	available ammonium, as N (mg/kg dw)
	available nitrate, as N (mg/kg dw)
	total Kjeldahl nitrogen (TKN) (% dw)
	total nitrogen (% dw)
	available phosphate, as P (mg/kg dw)
	available potassium (mg/kg dw)
	available sulphate, as S (mg/kg dw)
Total metals	aluminum (mg/kg dw)
	antimony (mg/kg dw)
	arsenic (mg/kg dw)
	barium (mg/kg dw)
	beryllium (mg/kg dw)
	bismuth (mg/kg dw)
	boron (mg/kg dw)
	cadmium (mg/kg dw)
	calcium (mg/kg dw)
	cesium (mg/kg dw)
	chromium (mg/kg dw)
	cobalt (mg/kg dw)
	copper (mg/kg dw)
	iron (mg/kg dw)
	lead (mg/kg dw)
	lithium (mg/kg dw)
	magnesium (mg/kg dw)
	manganese (mg/kg dw)
	mercury (mg/kg dw)
	molybdenum (mg/kg dw)
	nickel (mg/kg dw)
	phosphorus (mg/kg dw)
	potassium (mg/kg dw)
	rubidium (mg/kg dw)
	selenium (mg/kg dw)
	silver (mg/kg dw)
	sodium (mg/kg dw)
	strontium (mg/kg dw)
	thallium (mg/kg dw)
	tin (mg/kg dw)
	titanium (mg/kg dw)
	uranium (mg/kg dw)
	vanadium (mg/kg dw)
	zinc (mg/kg dw)

Table 4-2 2013 Sediment Quality Parameter List for Samples Collected in Snap Lake, Northeast Lake, and Lake 13

Parameter Group	Parameter [Units]
Polycyclic aromatic hydrocarbons (PAHs)	acenaphthene (mg/kg dw)
	acenaphthylene (mg/kg dw)
	acridine (mg/kg dw)
	anthracene (mg/kg dw)
	benzo[a]anthracene (mg/kg dw)
	benzo[a]pyrene (mg/kg dw)
	benzo[b&j]fluoranthene (mg/kg dw)
	benzo[g,h,i]perylene (mg/kg dw)
	benzo[k]fluoranthene (mg/kg dw)
	chrysene (mg/kg dw)
	dibenzo[a,h]anthracene (mg/kg dw)
	fluoranthene (mg/kg dw)
	fluorene (mg/kg dw)
	indeno[1,2,3-cd]pyrene (mg/kg dw)
	2-methylnaphthalene (mg/kg dw)
	naphthalene (mg/kg dw)
	phenanthrene (mg/kg dw)
	pyrene (mg/kg dw)
	quinoline (mg/kg dw)

% = percent; % dw = percent dry weight; mg/kg dw = milligrams per kilogram dry weight; N = nitrogen; P = phosphorus; S = sulphur.

Analyses for moisture, total metals, and PAHs were performed by the ALS Edmonton laboratory, and analyses for particle size, carbon content, and nutrients were performed by the ALS Saskatoon laboratory.

4.2.1.5 Supporting Environmental Variables

Supporting environmental information recorded during the 2013 sediment sampling program was:

- sampling date and time;
- weather conditions, such as air temperature and wind velocity;
- global positioning system (GPS) coordinates recorded as Universal Transverse Mercator (UTM) for each station;
- water depth; and,
- vertical profiles of water temperature, DO, pH, and conductivity, measured at 1-m intervals.

Station locations were identified using a hand-held Garmin GPS unit with UTM coordinates, in conjunction with topographical maps showing station locations. A YSI 650 Multiparameter Display System water quality meter with a YSI 600 Quick Sample multi-parameter water quality probe were used to measure water quality profiles. Details of the field water quality measurements are provided in Section 3.2.

4.2.2 Data Analyses

4.2.2.1 Approach

Sediment quality data analysis is designed to answer the key questions listed in Section 4.1.2. An overview of the analysis approach associated with these three questions is provided in Table 4-3. These three key questions are applicable in years when the full set of AEMP stations is monitored. In those years when sediment quality is only monitored at the diffuser station, as was the case for 2013, the sediment chemistry results for that station are compared to SQGs and temporal trends are assessed through comparison to data collected in previous years. Specific details relevant to data analysis methods to address the key questions are provided in Sections 4.2.2.3 to 4.2.2.5.

Table 4-3 Overview of Analysis Approach for Sediment Quality Key Questions

Key Question	Overview of Analysis Approach
1. Are concentrations of sediment quality parameters above or below SQGs?	Concentrations of sediment quality parameters are compared to appropriate SQGs. Instances where concentrations are above SQGs are identified and qualitatively assessed for potential Mine-related causes.
2. Are there differences in sediment quality in Snap Lake relative to reference lakes and, if so, are they related to the Mine?	Statistical tests (e.g., analysis of variance) are used to determine whether there are statistically significant differences in mean parameter concentrations between Snap Lake and the reference lakes.
3. Are concentrations of sediment quality parameters increasing over time?	Analyses of temporal patterns in concentrations of sediment quality parameters since 2004 baseline are performed using statistical tests (e.g., Mann-Kendall or other appropriate test) to quantify the statistical significance of any potential temporal trends. Mean parameter concentrations are compared to normal ranges.

SQG = sediment quality guideline; e.g. = for example.

4.2.2.2 Data Compilation and Summary

The 2013 sediment quality data for Snap Lake, Northeast Lake, and Lake 13 were summarized separately in terms of the whole-lake mean, median, minimum, maximum, and standard deviation (SD) for each parameter, where applicable. For the Snap Lake particle size and TOC data, similar summary statistics were calculated for each of the three lake areas: northwest arm; diffuser; and, main basin. Concentrations reported as less than their detection limit (DL) were replaced with values equal to half their DL prior to statistical analyses.

The top 5-cm and top 2-cm sediment quality data from the diffuser station were compared by calculating the relative percent difference (RPD) between concentrations in the two sample types:

$$\text{RPD} = (\text{top 5 cm} - \text{top 2 cm}) / [(\text{top 5 cm} + \text{top 2 cm})/2] \times 100 \qquad \text{[Equation 4-1]}$$

The RPD is the same formula used to compare results from field or laboratory duplicate analyses, and is a measure of analytical precision. Relative percent differences (RPDs) were calculated for each parameter for the SNP 02-20e station. For this comparison, a positive RPD indicated that the parameter concentration was higher in the top 5-cm sample than in the top 2-cm sample.

4.2.2.3 Comparison to Sediment Quality Guidelines

Sediment quality data were compared to the Interim Sediment Quality Guidelines (ISQGs) and Probable Effect Levels (PEL) developed by the Canadian Council of Ministers of the Environment (CCME 1999 with updates), which were available for 7 metals analyzed in the diffuser and Lake 13 sediment samples as well as 13 PAHs that were only analyzed in Lake 13 sediments (Table 4-4). The ISQG is the concentration of a substance below which an adverse effect on aquatic life is unlikely; the PEL is the concentration of a substance above which adverse effects are expected to occur frequently, but not always. In practice, the application of generic numerical guidelines has yielded a high percentage of false positives (Chapman and Mann 1999). The observation of a sediment concentration above the PEL value for a given parameter should not be interpreted as an indication that actual ecological harm has occurred or will occur, but rather that this is a possibility. Biological assessment, such as evaluation of the benthic invertebrate community (Section 6), is necessary to determine whether adverse ecological effects are actually occurring.

Table 4-4 Canadian Sediment Quality Guidelines for the Protection of Freshwater Aquatic Life

Parameter	Guidelines (mg/kg dw)	
	ISQG	PEL
Total Metals		
arsenic	5.9	17
cadmium	0.6	3.5
chromium	37.3	90
copper	35.7	197
lead	35	91.3
mercury	0.17	0.49
zinc	123	315
Polycyclic Aromatic Hydrocarbons (PAHs)		
acenaphthene	0.0067	0.089
acenaphthylene	0.0059	0.128
anthracene	0.0469	0.245
benzo[a]anthracene	0.0317	0.385
benzo[a]pyrene	0.0319	0.782
chrysene	0.057	0.862
dibenzo[a,h]anthracene	0.006	0.135
fluoranthene	0.111	2.36
fluorene	0.021	0.144
2-methylnapthalene	0.0202	0.201
naphthalene	0.0346	0.391
phenanthrene	0.0419	0.515
pyrene	0.0530	0.875

Source: CCME (1999 with updates).

CCME = Canadian Council of Ministers of the Environment; ISQG = Interim Sediment Quality Guideline; PEL = Probable Effect Level; mg/kg dw = milligrams per kilogram dry weight.

4.2.2.4 Evaluation of Spatial Patterns

Spatial patterns in Snap Lake sediment quality were not assessed in 2013 because only the diffuser station sediments were analyzed for the full suite of AEMP parameters. In years when the full set of AEMP stations are analyzed, spatial patterns in Snap Lake sediment quality are assessed by testing for statistically significant differences in mean parameter concentrations between the Snap Lake main basin and the reference lakes. Statistical analyses of the 2013 particle size and TOC data are provided in the benthic invertebrate chapter (Section 5).

4.2.2.5 Evaluation of Temporal Trends

To illustrate temporal trends in sediment quality in Snap Lake, data collected from 2004 to 2013 were plotted as lake area means (northwest arm, diffuser, and main basin). All available diffuser station data were plotted: both the top 2-cm and top 5-cm layers for 2012 and 2013; the top 5-cm layers for 2007 to 2011; and, the bulk samples for 2004 to 2006. The 2008 to 2013 means for Northeast Lake, and the 2012 and 2013 means for Lake 13, were also included for comparison. For 2013, particle size and TOC data were available for all lake areas, whereas nutrient and metals data were only available for the diffuser and Lake 13.

Statistical analyses were performed to identify statistically significant, defined as probability (*P*) less than 0.10, temporal trends in sediment chemistry concentrations at the diffuser station, using a non-parametric Mann-Kendall test (Gilbert 1987). Both increasing and decreasing temporal trends were identified

To evaluate whether sediment quality in Snap Lake has changed relative to baseline conditions, diffuser station concentrations were compared with baseline (2004) conditions expressed as normal ranges calculated for each parameter. For particle size, TOC, total Kjeldahl nitrogen (TKN; which is a measure of organic nitrogen plus ammonia that does not include nitrate and nitrite), total nitrogen, and metals, normal ranges were expressed as the mean ±2SD calculated from the 2004 baseline sediment chemistry data for each parameter. Available nitrate, available phosphate, available potassium, and available sulphate were added to the target parameter list in 2005, and available ammonium was added in 2006; therefore, 2004 baseline data were not available. Normal ranges for these additional nutrients were calculated using data collected during the first year of monitoring, but only from stations with bottom conductivity less than 50 microSiemens per centimetre (µS/cm). These normal ranges were included in the time-series plots for each sediment quality parameter.

4.3 Quality Assurance and Quality Control

4.3.1 Overview of Procedures

Quality assurance and quality control (QA/QC) procedures govern all aspects of the AEMP (De Beers 2005), including field methods, laboratory analyses, data management, and reporting. Details of QA/QC procedures and results for the 2013 sediment quality samples are provided in Appendix 4A.

4.3.2 Summary of Results

In 2013, qualifiers were assigned to sediment sample data for DL increases for available ammonium, available nitrate, and available phosphate due to interference from sample matrix effects and for all PAHs due to high moisture content in samples. With two exceptions, these failures to meet data quality objectives (DQOs) were relatively minor and not expected to adversely affect data quality. The DLs used for available nitrate since 2010 were higher than in previous years, which made temporal comparisons difficult because more recent data were reported as undetected. The DLs for PAHs were increased by ten times due to high moisture content. Although PAHs were undetected in all the Lake 13 samples, the adjusted DLs for nine PAHs were above their respective CCME ISQGs. The specified DLs were met for most other parameters; where they were not met, DLs were low enough relative to concentrations measured in sediment samples that this did not adversely affect data quality.

Sample holding times were met for all analyses, except for carbon analyses performed on all the Snap Lake samples and one Northeast Lake sample. However, samples were kept cool and dark in sealed containers and it is unlikely that data quality was affected. All of the requested analyses were performed, except that moisture content was not determined for the Northeast Lake or Snap Lake samples (except for SNAP03).

Results from analyses of laboratory duplicates, laboratory reference materials, and laboratory method blanks met their respective DQOs.

Field duplicate samples were collected at one Snap Lake station and one Lake 13 station in 2013, using separately collected sets of grab samples. The purpose of the field duplicates was to evaluate sample variability. Comparison of each field duplicate to its original sample showed that there was general agreement in terms of measured parameter concentrations, except for available ammonium and antimony at one station, and available phosphate and titanium at the other station.

4.4 Results

4.4.1 Supporting Environmental Variables

Water depths at sediment sampling stations ranged from 10.5 to 15 m at Snap Lake stations, except for the deeper diffuser station SNP 02-20e (30.5 m). Water depths ranged from 10.2 to 13.5 m in Northeast Lake, and from 9.9 to 14.5 m in Lake 13.

4.4.2 Summary of 2013 Sediment Quality Data

The 2013 raw sediment quality data for Snap Lake, Northeast Lake, and Lake 13 are provided in Appendix 4B, Table 4B-1. All sediment quality data are reported on a dry weight (dw) basis, except for moisture content. Whole-lake means and summary statistics for particle size and carbon analyses are presented in Table 4-5 for Snap Lake, Northeast Lake, and Lake 13 sediments. Summarized chemistry data for nutrients, metals, and PAHs analyzed in sediments from the Snap Lake diffuser station and Lake 13 stations are presented in Table 4-6.

Sediments from the Snap Lake, Northeast Lake, and Lake 13 stations were comprised primarily of fine-grained silt and clay, with smaller amounts of sand. The percentage of fines ranged from 95% to 99% dw.

Total organic carbon (TOC) concentrations ranged from 12% to 22% dw in Snap Lake sediments, with all but three samples having TOC concentrations less than 18% dw. Sediments from Northeast Lake stations had TOC concentrations of 15% to 18% dw. Sediment TOC concentrations at Lake 13 stations were the lowest, ranging from 8% to 10% dw.

Table 4-5 Summary of 2013 Sediment Particle Size and Carbon Data for Snap Lake, Northeast Lake, and Lake 13, as Whole Lake Means and Summary Statistics

		Snap Lake Top 5-cm Sediment Samples						Northeast Lake Top 5-cm Sediment Samples						Lake 13 Top 5-cm Sediment Samples					
Parameter	Units (dw)	n	Mean	SD	Median	Min	Max	n	Mean	SD	Median	Min	Max	n	Mean	SD	Median	Min	Max
Physical																			
Moisture	%	2	94.4	0.28	94.4	94.2	94.6	0	-	-	-	-	-	5	91.3	1.3	91.6	89.9	93.0
% Gravel (>2 mm)	% dw	11	<0.1	0.00	<0.1	<0.1	<0.1	5	<0.1	0	<0.1	<0.1	<0.1	5	<0.1	0	<0.1	<0.1	<0.1
% Sand (2.0 mm - 0.063 mm)	% dw	11	1.8	1.21	1.72	0.28	4.46	5	2.5	1.7	3.6	0.5	4.2	5	1.9	1.3	1.2	0.7	3.9
% Silt (0.063 mm - 0.004 mm)	% dw	11	78.4	4.28	78.9	72.4	87.5	5	76.9	1.7	76.4	75.3	79.3	5	76.3	3.1	75.3	73.9	81.5
% Clay (<0.004 mm)	% dw	11	19.7	4.52	19.6	9.81	26.4	5	20.6	1.9	20.3	18.5	23.8	5	21.8	3.5	22.2	17.3	25.2
% Fines (Silt + Clay)	% dw	11	98.1	1.21	98.3	95.5	99.7	5	97.5	1.7	96.4	95.9	99.6	5	98.1	1.3	98.8	96.1	99.3
Inorganic / Organic Carbon																			
Total Carbon by Combustion	% dw	11	18.2	2.79	18.1	11.8	22.2	5	15.9	1.3	15.6	14.9	18.2	5	9.3	1.3	9.6	7.8	10.5
Inorganic Carbon	% dw	11	0.06	0.02	<0.1	<0.1	0.11	5	<0.1	0.0	<0.1	<0.1	<0.1	5	0.1	0.0	0.1	0.1	0.1
Total Organic Carbon	% dw	11	18.2	2.78	18.1	11.8	22.2	5	15.9	1.3	15.6	14.9	18.2	5	9.2	1.3	9.5	7.7	10.4

- = not available/not applicable; < = less than; > = greater than; n = sample size; Min = minimum; Max = maximum; SD = standard deviation; cm = centimetre; mm = millimetre; % dw = percent dry weight; % = percent.

Table 4-6 Summary of 2013 Sediment Chemistry Data for Snap Lake Diffuser and Lake 13 Stations

Parameter	Units (dw)	Diffuser (SNP 02-20e)		Lake 13 Top 5-cm Sediment Samples					
		Top 5-cm	Top 2-cm	n	Mean	SD	Median	Minimum	Maximum
Nutrients									
Available Ammonium-N	mg/kg dw	5.8	29.4	5	31.3	7.7	30.6	20.0	41.2
Available Nitrate-N	mg/kg dw	<6	<6	5	<6	0	<6	<6	<6
Total Kjeldahl Nitrogen (TKN)	% dw	1.34	1.43	5	0.75	0.09	0.80	0.64	0.84
Total Nitrogen	% dw	1.36	1.48	5	0.75	0.11	0.78	0.63	0.85
Available Phosphate-P	mg/kg dw	57.1	34.5	5	31.9	34.2	18.6	<2	86.6
Available Potassium	mg/kg dw	124	207	5	139	42	119	115	213
Available Sulfate-S	mg/kg dw	167	289	5	83.2	96.0	35.3	21.4	251
Metals									
Aluminum	mg/kg dw	17,900	17,400	5	21,240	3,774	20,800	16,100	26,100
Antimony	mg/kg dw	0.15	0.25	5	0.10	0.05	0.12	<0.10	0.15
Arsenic	mg/kg dw	3.11	2.99	5	24.0	27.5	14.1	6.3	72.8
Barium	mg/kg dw	94.3	83.9	5	561	944	140	115	2,250
Beryllium	mg/kg dw	0.80	0.83	5	0.84	0.13	0.89	0.62	0.94
Bismuth	mg/kg dw	0.59	0.61	5	0.67	0.10	0.66	0.55	0.79
Boron	mg/kg dw	18.3	20.2	5	7.68	1.84	7.50	5.50	10.0
Cadmium	mg/kg dw	0.47	0.44	5	0.41	0.13	0.43	0.26	0.59
Calcium	mg/kg dw	5,140	7,830	5	2,682	348	2,790	2,140	3,070
Cesium	mg/kg dw	1.74	1.73	5	2.96	0.65	3.12	2.10	3.77
Chromium	mg/kg dw	45.1	41.0	5	53.7	10.9	57.0	37.0	65.8
Cobalt	mg/kg dw	12.1	14.1	5	34.6	30.7	20.4	15.4	88.6
Copper	mg/kg dw	105	101	5	66.3	7.4	68.3	56.4	73.8
Iron	mg/kg dw	25,400	28,400	5	65,300	50,244	48,900	31,700	154,000
Lead	mg/kg dw	7.16	9.26	5	7.36	0.74	7.39	6.33	8.10
Lithium	mg/kg dw	22.8	27.9	5	37.6	9.7	39.7	24.4	49.6
Magnesium	mg/kg dw	4,610	7,350	5	6,450	1,326	6,510	4,590	8,180
Manganese	mg/kg dw	254	509	5	14,325	28,029	1,820	585	64,400
Mercury	mg/kg dw	0.065	0.082	5	0.030	0.012	<0.050	<0.050	0.051
Molybdenum	mg/kg dw	7.91	9.92	5	7.90	3.42	6.36	5.28	13.4
Nickel	mg/kg dw	40.0	52.5	5	55.8	14.1	53.1	41.6	74.8
Phosphorus	mg/kg dw	1,610	1,590	5	1,324	407	1,140	961	1,970
Potassium	mg/kg dw	1,600	1,530	5	3,062	677	3,210	2,220	4,010
Rubidium	mg/kg dw	13.0	12.5	5	21.7	4.4	21.9	17.7	28.3
Selenium	mg/kg dw	1.48	1.30	5	0.82	0.15	0.85	0.62	1.00

Table 4-6 Summary of 2013 Sediment Chemistry Data for Snap Lake Diffuser and Lake 13 Stations

Parameter	Units (dw)	Diffuser (SNP 02-20e)		Lake 13 Top 5-cm Sediment Samples					
		Top 5-cm	Top 2-cm	n	Mean	SD	Median	Minimum	Maximum
Silver	mg/kg dw	0.27	0.33	5	<0.20	0	<0.20	<0.20	<0.20
Sodium	mg/kg dw	520	770	5	160	22	160	130	190
Strontium	mg/kg dw	66.7	125	5	25.0	7.5	21.5	20.2	38.2
Thallium	mg/kg dw	0.064	0.159	5	0.29	0.11	0.24	0.19	0.47
Tin	mg/kg dw	<2	<2	5	<2	0	<2	<2	<2
Titanium	mg/kg dw	224	203	5	415	116	443	295	569
Uranium	mg/kg dw	8.14	8.29	5	6.86	1.97	6.66	4.70	10.0
Vanadium	mg/kg dw	33.1	32.1	5	48.0	8.2	48.5	35.2	57.3
Zinc	mg/kg dw	126	111	5	112	9	109	101	123
Polycyclic Aromatic Hydrocarbons (PAHs)									
Acenaphthene	mg/kg dw	-	-	5	<0.050	0	<0.050	<0.050	<0.050
Acenaphthylene	mg/kg dw	-	-	5	<0.050	0	<0.050	<0.050	<0.050
Acridine	mg/kg dw	-	-	5	<0.050	0	<0.050	<0.050	<0.050
Anthracene	mg/kg dw	-	-	5	<0.040	0	<0.040	<0.040	<0.040
Benzo(a)anthracene	mg/kg dw	-	-	5	<0.050	0	<0.050	<0.050	<0.050
Benzo(a)pyrene	mg/kg dw	-	-	5	<0.050	0	<0.050	<0.050	<0.050
Benzo(b&j)fluoranthene	mg/kg dw	-	-	5	<0.050	0	<0.050	<0.050	<0.050
Benzo(g,h,i)perylene	mg/kg dw	-	-	5	<0.050	0	<0.050	<0.050	<0.050
Benzo(k)fluoranthene	mg/kg dw	-	-	5	<0.050	0	<0.050	<0.050	<0.050
Chrysene	mg/kg dw	-	-	5	<0.050	0	<0.050	<0.050	<0.050
Dibenzo(a,h)anthracene	mg/kg dw	-	-	5	<0.050	0	<0.050	<0.050	<0.050
Fluoranthene	mg/kg dw	-	-	5	<0.050	0	<0.050	<0.050	<0.050
Fluorene	mg/kg dw	-	-	5	<0.050	0	<0.050	<0.050	<0.050
Indeno(1,2,3-cd)pyrene	mg/kg dw	-	-	5	<0.050	0	<0.050	<0.050	<0.050
2-Methylnaphthalene	mg/kg dw	-	-	5	<0.050	0	<0.050	<0.050	<0.050
Naphthalene	mg/kg dw	-	-	5	<0.050	0	<0.050	<0.050	<0.050
Phenanthrene	mg/kg dw	-	-	5	<0.050	0	<0.050	<0.050	<0.050
Pyrene	mg/kg dw	-	-	5	<0.050	0	<0.050	<0.050	<0.050
Quinoline	mg/kg dw	-	-	5	<0.050	0	<0.050	<0.050	<0.050

- = not applicable / not available; < = less than; n = sample size; SD = standard deviation; N = nitrogen; P = phosphorus; S = sulphur; cm = centimetre; % dw = percent dry weight; mg/kg dw = milligrams per kilogram dry weight.

Nutrients and metals were only analyzed in sediments from the Snap Lake diffuser station and the five Lake 13 stations. Available nitrate, less than 6.0 milligrams per kilogram dry weight (mg/kg dw), and tin, less than 2.0 mg/kg dw, were undetected in all diffuser station and Lake 13 samples in 2013. Mercury was detected in the diffuser station samples; it was only detected in one Lake 13 sample, at a concentration close to the DL (0.051 versus 0.050 mg/kg dw). Silver was detected in the diffuser station samples, but not in the Lake 13 samples.

Concentrations of most of the 39 nutrients and metals detected in Lake 13 sediments varied by less than a factor of three among individual stations; parameters with larger concentration ranges were available phosphate, available sulphate, arsenic, barium, cobalt, iron, and manganese. One Lake 13 station (LK13-03) had considerably higher concentrations of arsenic, barium, cobalt, iron, and manganese than the other four Lake 13 stations, which accounted for the larger concentration ranges for those parameters. Similar results were obtained for the LK13-03 station in 2012, indicating that sediment quality in this northeast area of Lake 13 differs from elsewhere in the lake.

Analyses of PAH concentrations in Lake 13 sediments were included in 2013 to provide data to support development of the winter road study design. None of the individual PAHs were detected (less than 0.050 mg/kg dw except less than 0.040 mg/kg dw for anthracene) in samples prepared by compositing the top 5-cm layer of sediment at each station.

Mean parameter concentrations reported for Lake 13 sediments were compared to concentrations reported for the top 5-cm layer sample from the Snap Lake diffuser station. Of the 39 nutrients and metals detected in sediment samples in 2013, mean concentrations of 21 parameters were higher in Lake 13 than in the diffuser station sample. These parameters are identified in Table 4-7; the same parameters were also higher in Lake 13 in 2012.

The top 5-cm layer of sediment is currently sampled for AEMP sediment quality monitoring, and has been since 2007. However, because sedimentation rates in arctic lakes are known to be low and concerns have been expressed as to whether the top 5-cm layer is too thick to be representative of recent Mine-related deposition, monitoring of both the top 2-cm and top 5-cm layers of sediment has been performed at the Snap Lake diffuser station since 2012 and is now an annual monitoring component of the AEMP. Results for both 2012 and 2013 are provided in Table 4-8, and time-series plots are provided in Appendix 4B, Figure 4B-1. For each parameter, RPDs were calculated to provide a measure of the difference in concentrations between the two sampling depths (Table 4-8). Relative percent differences (RPDs) are a measure typically used to assess analytical precision through comparison of laboratory duplicate samples, with an RPD that is less than or equal to 20% representing good agreement between a sample and its corresponding laboratory duplicate. For this sampling depth comparison, the differences between parameter concentrations for the two sampling depths would need to be larger than the amount of variability that typically occurs between laboratory duplicate samples for the differences in concentrations to be considered meaningful.

Table 4-7 Occurrence of Maximum Parameter Concentrations in Snap Lake and Lake 13 in 2013

Parameters Having Maximum Concentrations at Snap Lake Diffuser Station	Parameters Having Maximum Mean Concentrations in Lake 13
Total Kjeldahl nitrogen (TKN)	Available ammonium, as N
Total nitrogen	Available potassium
Available phosphate, as P	Aluminum
Available sulphate, as S	Arsenic
Antimony	Barium
Boron	Beryllium
Cadmium	Bismuth
Calcium	Cesium
Copper	Chromium
Mercury	Cobalt
Molybdenum	Iron
Phosphorus	Lead
Selenium	Lithium
Silver	Magnesium
Sodium	Manganese
Strontium	Nickel
Uranium	Potassium
Zinc	Rubidium
	Thallium
	Titanium
	Vanadium

N = nitrogen; P = phosphorus; S = sulphate.

Table 4-8 Differences in Sediment Chemistry for Snap Lake Diffuser Station Samples Collected in 2012 and 2013

Sampling Station		Diffuser Station SNP 02-20e (2013)			Diffuser Station SNP 02-20e (2012)		
Sediment Depth (cm)	Units	Top 5-cm	Top 2-cm	RPD	Top 5-cm	Top 2-cm	RPD
Physical/Carbon							
Fines (Silt + Clay)	% dw	97.3	96.8	1%	97.9	98.7	-1%
Total Organic Carbon	% dw	17.9	18.1	-1%	16.5	18.4	-11%
Nutrients							
Available Ammonium, as N	mg/kg dw	5.8	29.4	-134%	65	<25	89%
Total Kjeldahl Nitrogen (TKN)	% dw	1.34	1.43	-6%	1.33	1.51	-13%
Total Nitrogen	% dw	1.36	1.48	-8%	1.28	1.54	-18%
Available Nitrate, as N	mg/kg dw	<6.0	<6.0	0%	44.0	10.4	124%
Available Phosphate, as P	mg/kg dw	57.1	34.5	49%	426	14.1	187%
Available Potassium	mg/kg dw	124	207	-50%	1,320	269	132%
Available Sulphate, as S	mg/kg dw	167	289	-54%	59.3	253	-124%
Metals							
Aluminum	mg/kg dw	17,900	17,400	3%	11,500	11,000	4%
Antimony	mg/kg dw	0.15	0.25	-50%	0.12	0.38	-104%
Arsenic	mg/kg dw	3.11	2.99	4%	1.53	3.13	-69%
Barium	mg/kg dw	94.3	83.9	12%	68.9	76.8	-11%
Beryllium	mg/kg dw	0.8	0.83	-4%	0.69	0.50	32%
Bismuth	mg/kg dw	0.59	0.61	-3%	0.68	0.77	-12%
Boron	mg/kg dw	18.3	20.2	-10%	20.1	26.0	-26%
Cadmium	mg/kg dw	0.47	0.44	7%	0.45	0.44	2%
Calcium	mg/kg dw	5,140	7,830	-41%	3,930	6,490	-49%
Cesium	mg/kg dw	1.74	1.73	1%	1.75	1.67	5%
Chromium	mg/kg dw	45.1	41	10%	30.8	38.7	-23%
Cobalt	mg/kg dw	12.1	14.1	-15%	11.1	15.4	-32%
Copper	mg/kg dw	105	101	4%	106	94.8	11%
Iron	mg/kg dw	25,400	28,400	-11%	17,600	26,300	-40%
Lead	mg/kg dw	7.16	9.26	-26%	5.33	10.4	-64%
Lithium	mg/kg dw	22.8	27.9	-20%	20.4	22.9	-12%
Magnesium	mg/kg dw	4,610	7,350	-46%	2,920	5,790	-66%
Manganese	mg/kg dw	254	509	-67%	246	373	-41%
Mercury	mg/kg dw	0.065	0.082	-23%	0.062	0.101	-48%
Molybdenum	mg/kg dw	7.91	9.92	-23%	9.18	13.5	-38%
Nickel	mg/kg dw	40	52.5	-27%	33.1	53.9	-48%
Phosphorus	mg/kg dw	1,610	1,590	1%	1,510	1,620	-7%
Potassium	mg/kg dw	1,600	1,530	4%	1,380	1,470	-6%
Rubidium	mg/kg dw	13	12.5	4%	13.3	13.0	2%
Selenium	mg/kg dw	1.48	1.3	13%	1.77	2.03	-14%
Silver	mg/kg dw	0.27	0.33	-20%	0.23	0.39	-52%
Sodium	mg/kg dw	520	770	-39%	440	810	-59%
Strontium	mg/kg dw	66.7	125	-61%	44.9	110	-84%
Thallium	mg/kg dw	0.064	0.159	-85%	0.135	0.127	6%
Titanium	mg/kg dw	224	203	10%	245	248	-1%
Uranium	mg/kg dw	8.14	8.29	-2%	8.27	8.80	-6%
Vanadium	mg/kg dw	33.1	32.1	3%	29.9	29.6	1%
Zinc	mg/kg dw	126	111	13%	110	102	8%

RPD = relative percent difference; N = nitrogen; P = phosphate; S = sulphate; cm = centimetre; < = less than; % = percent; % dw = percent dry weight; mg/kg dw = milligrams per kilogram dry weight.

In 2013, percent fines and TOC concentrations at both diffuser station sampling depths were similar with RPDs of 1% and -1%, respectively. Of the 39 nutrients and metals detected in the 2013 diffuser station samples, 14 parameters had RPDs that were greater than 20% and negative, which in this case meant that the concentration in the top 2-cm layer was higher than in the top 5-cm layer: available ammonium; available potassium; available sulphate; antimony; calcium; lead; magnesium; manganese; mercury; molybdenum; nickel; sodium; strontium; and, thallium. Available ammonium had the largest negative RPD (-134%) in 2013, with concentrations of 5.8 milligrams per kilogram (mg/kg) in the top 5-cm layer and 29.4 mg/kg in the top 2-cm layer. Available ammonium concentrations were variable at this station, as indicated by the variability in field duplicate results for the top 5-cm layer (5.8 versus 23.2 mg/kg); however, even if the higher field duplicate result were used for the sampling depth comparison, the RPD would be -24%. Only available phosphate had an RPD greater than 20% that was positive.

Nutrient and metal parameters having concentrations at the two sampling depths that resulted in negative RPDs greater than 20% in both 2012 and 2013 were: available sulphate; antimony; calcium; lead; magnesium; manganese; mercury; molybdenum; nickel; sodium; and, strontium. Although data were only available for two years, results indicate that concentrations of these parameters may be consistently higher in more recently deposited sediments at the Snap Lake diffuser station and therefore influenced by exposure to treated effluent. Available phosphate was the only parameter to have a positive RPD greater than 20% in both 2012 and 2013, indicating that concentrations were lower in more recently deposited sediments.

4.4.3 Comparison to Sediment Quality Guidelines

Of the parameters analyzed in Snap Lake and/or Lake 13 sediment samples in 2013, Canadian SQGs were available for 7 metals and 13 PAHs (Table 4-4). Concentrations of a number of those metals were above SQGs in Snap Lake and Lake 13 sediments in 2013, as was observed in previous years (Table 4-9). Data from both Snap Lake diffuser station sampling depths were compared to SQGs. Because the number of stations sampled varied since the early years of the AEMP, comparisons to each SQG are presented both in terms of numbers of stations with concentrations above the SQG as well as percentage occurrence.

Arsenic concentrations at the Snap Lake diffuser station were below the ISQG in 2013. Arsenic concentrations in Snap Lake have only occasionally been above the ISQG since 2004 (Table 4-9); since 2007, exceedance of the arsenic ISQG has only occurred at SNAP20 in the northwest arm, and only in 2007, 2009, and 2011.

Cadmium concentrations at the Snap Lake diffuser station were below the ISQG in 2013. Although cadmium concentrations elsewhere in Snap Lake have exceeded the ISQG in previous years, concentrations at the Snap Lake diffuser station have been at or below the ISQG since 2005.

Table 4-9 Sediment Quality Guideline Exceedances for Metals in Snap Lake, 2004 to 2013, and Lake 13, 2012 to 2013

Lake	Year/Sampling Method	Guideline	n	Arsenic	Cadmium	Chromium	Copper	Lead	Mercury	Zinc
Snap Lake	2004 Bulk	ISQG	12	-	6/12 (50%)	5/12 (42%)	12/12 (100%)	-	-	12/12 (100%)
		PEL	12	-	-	-	-	-	-	1/12 (8%)
	2005 Bulk	ISQG	17	2/17 (12%)	5/17 (29%)	7/17 (41%)	13/17 (76%)	2/17 (12%)	-	8/17 (47%)
		PEL	17	-	-	-	-	2/17 (12%)	-	-
	2006 Bulk	ISQG	18	1/18 (6%)	9/18 (50%)	9/18 (50%)	18/18 (100%)	-	-	17/18 (94%)
		PEL	18	-	-	-	-	-	-	-
	2007 Top 5-cm	ISQG	17	1/13 (8%)	13/17 (76%)	4/17 (24%)	17/17 (100%)	-	-	16/17 (94%)
		PEL	17	-	-	-	-	-	-	-
	2008 Top 5-cm	ISQG	18	-	12/18 (67%)	2/18 (11%)	18/18 (100%)	-	-	15/18 (83%)
		PEL	18	-	-	-	-	-	-	-
	2009 Top 5-cm	ISQG	18	1/18 (6%)	10/18 (56%)	-	18/18 (100%)	-	-	13/18 (72%)
		PEL	18	-	-	-	-	-	-	-
	2010 Top 5-cm	ISQG	18	-	8/18 (44%)	5/18 (28%)	17/18 (94%)	-	-	13/18 (72%)
		PEL	18	-	-	-	-	-	-	-
	2011 Top 5-cm	ISQG	18	1/18 (6%)	5/18 (28%)	5/18 (28%)	18/18 (100%)	-	-	12/18 (67%)
		PEL	18	-	-	-	-	-	-	-
	2012 Top 5-cm	ISQG	18	-	10/18 (56%)	4/18 (22%)	18/18 (100%)	-	-	13/18 (72%)
		PEL	18	-	-	-	-	-	-	-
	2013 Top 2 and 5-cm	ISQG	2	-	-	2/2 (100%)	2/2 (100%)	-	-	1/2 (50%)
		PEL	2	-	-	-	-	-	-	-
Lake 13	2012 Top 5-cm	ISQG	5	4/5 (80%)	1/5 (20%)	5/5 (100%)	5/5 (100%)	-	-	-
		PEL	5	2/5 (40%)	-	-	-	-	-	-
	2013 Top 5-cm	ISQG	5	5/5 (100%)	-	4/5 (80%)	5/5 (100%)	-	-	-
		PEL	5	1/5 (20%)	-	-	-	-	-	-

Notes: Percentage in parentheses indicates the percentage of stations where the sediment concentration was above the relevant guideline. Number before the "/" indicates the number of stations where the sediment concentration was above the relevant guideline, and the number after the "/" indicates the number of stations sampled (or number of stations for which data were available).

n = sample size; - = no stations had sediment concentrations exceeding the guideline; ISQG= Interim Sediment Quality Guideline; PEL= Probable Effect Level; cm = centimetre; % = percent.

Chromium and copper concentrations were above their respective ISQGs in both Snap Lake diffuser station samples in 2013. Although chromium concentrations elsewhere in Snap Lake have exceeded the ISQG in previous years, concentrations at the Snap Lake diffuser station were only above the ISQG in 2006, 2012, and 2013. Copper concentrations have been above the ISQG in other areas of Snap Lake in previous years, and at the diffuser station in all years except 2005.

Lead concentrations have been below the ISQG at all Snap Lake stations since 2004, with the exception of anomalously high results in 2005 for two diffuser and near-field stations. Those lead concentrations of 161 and 373 mg/kg dw were much higher than any other lead concentrations reported for Snap Lake, which are typically less than 10 mg/kg dw. Concentrations of mercury in Snap Lake sediments have been below the ISQG at all stations in all years.

Zinc concentrations in the Snap Lake diffuser station samples were above the ISQG in the top 5-cm sample, and below the ISQG in the top 2-cm sample, in 2013. Although zinc concentrations elsewhere in Snap Lake have exceeded the ISQG in previous years, including all the 2004 baseline samples, concentrations at the Snap Lake diffuser station were only above the ISQG in 2006, 2007, and 2013.

In Lake 13 sediments, concentrations of lead, mercury, and zinc were below their respective ISQGs at all five stations in 2012 and 2013, and cadmium was only above its ISQG at one station in 2012. Concentrations of chromium and copper were above their ISQGs for at least four of the five stations in 2012 and 2013; this was consistent with observations for Snap Lake and Northeast Lake in previous years and suggests that concentrations are naturally elevated in sediments in the area surrounding the Mine. In 2013, arsenic concentrations were higher in Lake 13 sediments (6.3 to 72.8 mg/kg dw) than at the Snap Lake diffuser station (2.99 and 3.11 mg/kg dw); concentrations were above the arsenic ISQG at all five Lake 13 stations and above the PEL at one station. Similar results were reported for arsenic concentrations in Lake 13 sediments in 2012; in both years the concentration at the LK13-03 station was considerably higher than at the other four stations. Arsenic concentrations measured in three Lake 13 sediment samples in July 2005 ranged from 4.0 to 6.2 mg/kg dw (Golder 2005). However, the EAR reported maximum arsenic sediment concentrations in the Lockhart River watershed of 49.0 mg/kg dw in 1993/1994 and 55.3 mg/kg dw in August 1999 (De Beers 2002). Thus, the Lake 13 arsenic sediment concentrations are consistent with natural variability.

Concentrations of PAHs were measured in Lake 13 sediments in 2013 to support development of the winter road study design. Polycyclic aromatic hydrocarbons (PAHs) were undetected at all stations (less than 0.050 mg/kg dw, except less than 0.04 mg/kg dw for anthracene), but because the DLs had to be increased by ten times due to high moisture content in the samples the adjusted DLs for nine individual PAHs were above their respective ISQGs: acenaphthene; acenapthylene; benzo[a]anthracene; benzo[a]pyrene; dibenzo[a,h]anthracene; fluorene; 2-methylnaphthalene; naphthalene; and, phenanthrene. Although unlikely, there is uncertainty as to whether these SQGs may have been exceeded.

4.4.4 Temporal Trends in Snap Lake Sediment Quality

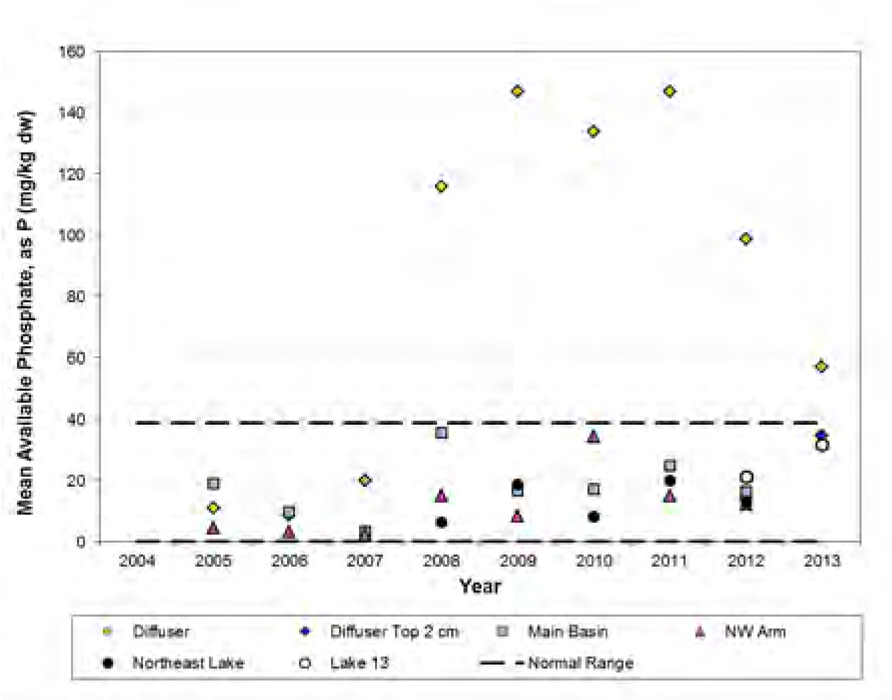
Temporal trends in sediment quality were only assessed statistically for the Snap Lake diffuser station in 2013, using data from bulk samples and top 5-cm layer samples collected between 2005 and 2013. Temporal trends were not assessed for the diffuser station top 2-cm layer, or for Lake 13 sediments, because only two years of data were available. Time-series plots for all sediment quality parameters are provided in Appendix 4B, Figure 4B-1; for those parameters with SQGs, the ISQG is shown on the plot as the benchmark. For comparison, the plots also show mean concentrations for other areas of Snap Lake and Northeast Lake from previous years.

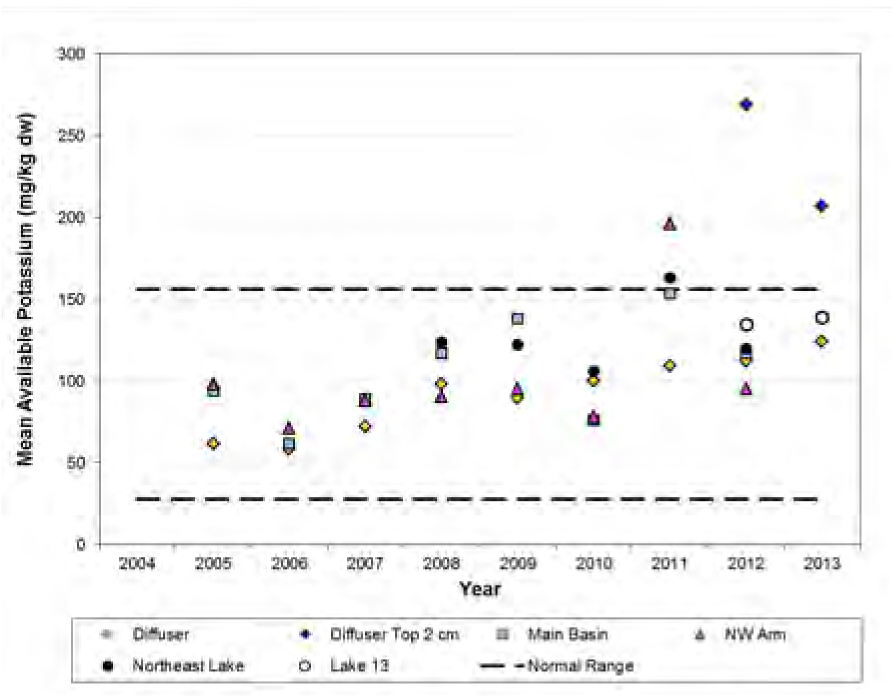
Results of the Mann-Kendall statistical analyses for temporal trends showed that 27 of the 39 parameters monitored at the diffuser area, and included for statistical trend analyses, had increasing trends between 2005 and 2013, although these trends were not all statistically significant. Baseline 2004 data were not available for the diffuser area. Of those 27 parameters, 12 had statistically significant increasing trends ($P<0.10$): available potassium; available sulphate; aluminum; boron; calcium; iron; mercury; molybdenum; selenium; silver; sodium; and, strontium. Four parameters had statistically significant decreasing trends ($P<0.10$) at the Snap Lake diffuser station: barium; cesium; thallium; and, titanium. The historical patterns of mean area concentrations for the parameters showing statistically significant positive trends at the Snap Lake diffuser station over the period from 2005 to 2013 are illustrated in Figure 4-4; mean concentrations for Northeast Lake and Lake 13 are also included for comparison.

Concentrations of parameters measured in sediments at the Snap Lake diffuser station in 2013 were compared to their normal ranges, estimated as the baseline whole-lake mean ±2SD (Table 4-10) for each parameter. Table 4-10 also shows annual comparisons of the Snap Lake area mean concentrations for the northwest arm, diffuser, and main basin areas to normal ranges from 2005 to 2012. Concentrations of available potassium, available phosphate, available sulphate, antimony, calcium, lead, magnesium, manganese, mercury, selenium, silver, sodium, and strontium were above their respective normal ranges in 2013. Antimony, selenium, and silver were not detected in 2004 baseline sediment samples; therefore, their normal ranges are equal to their respective DLs. Time-series plots for parameters exceeding their normal ranges are provided in Figure 4-4.

The magnitude and pattern of some of these statistically significant temporal trends also need to be considered. Small incremental increases in concentration from year to year can result in a statistically significant temporal trend being identified, such as for available potassium and iron, even though the overall net change in concentration is small and unlikely to result in significant adverse effects to biota associated with the sediments. Similarly, concentrations of several parameters at the diffuser area increased markedly between 2005 and 2007 but have remained relatively consistent since then. The marked gradient in available phosphate concentrations observed in the diffuser area from 2007 to 2011 decreased in 2012 and 2013.

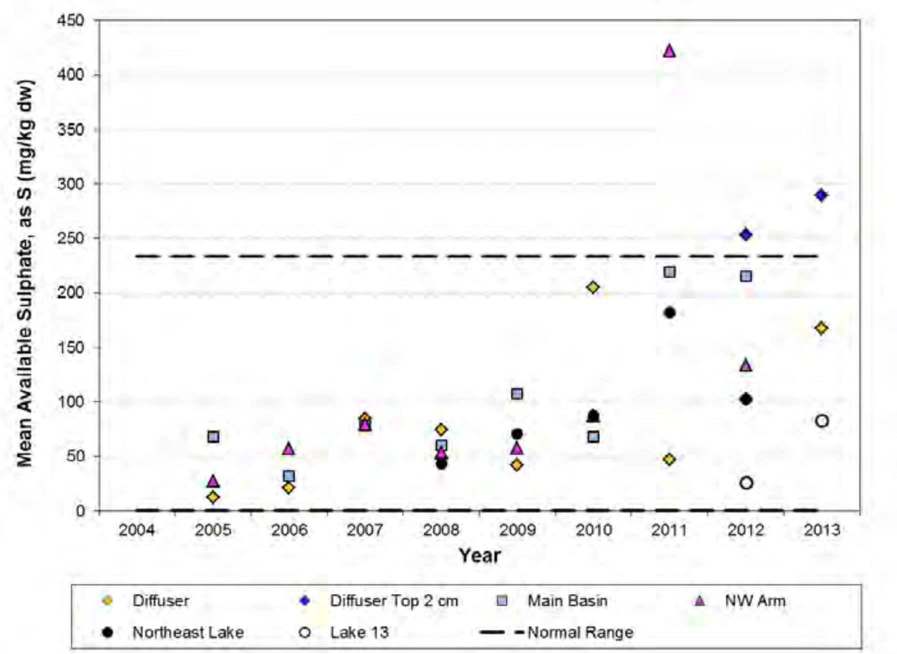
Figure 4-4 Time-Series Plots for Parameters With Statistically Significant Positive Temporal Trends or Concentrations Outside Normal Range at Snap Lake Diffuser Station, 2004 to 2013

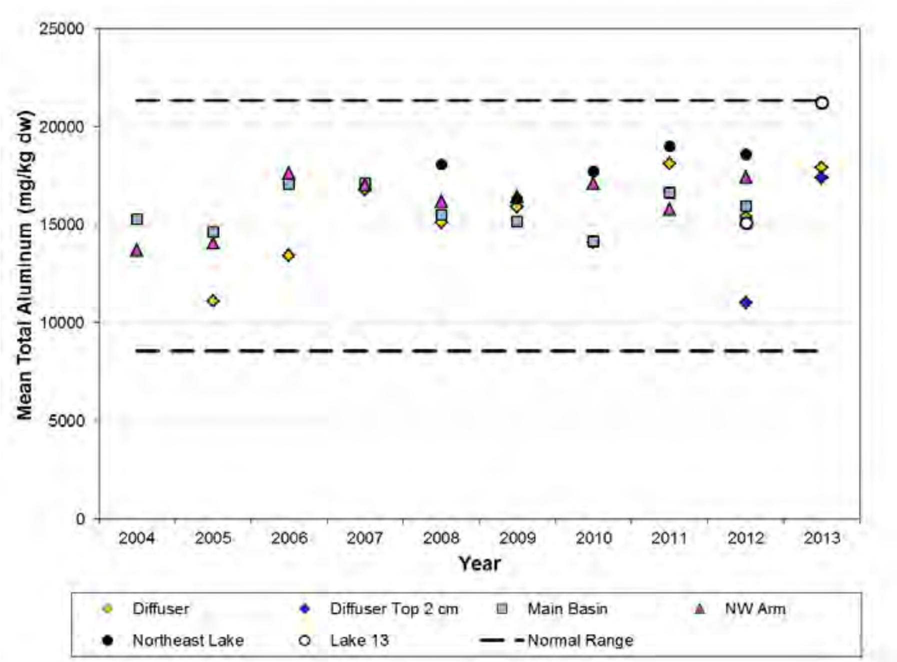




mg/kg dw = milligrams per kilogram dry weight; P = phosphorus; cm = centimetre; NW Arm = Northwest Arm.

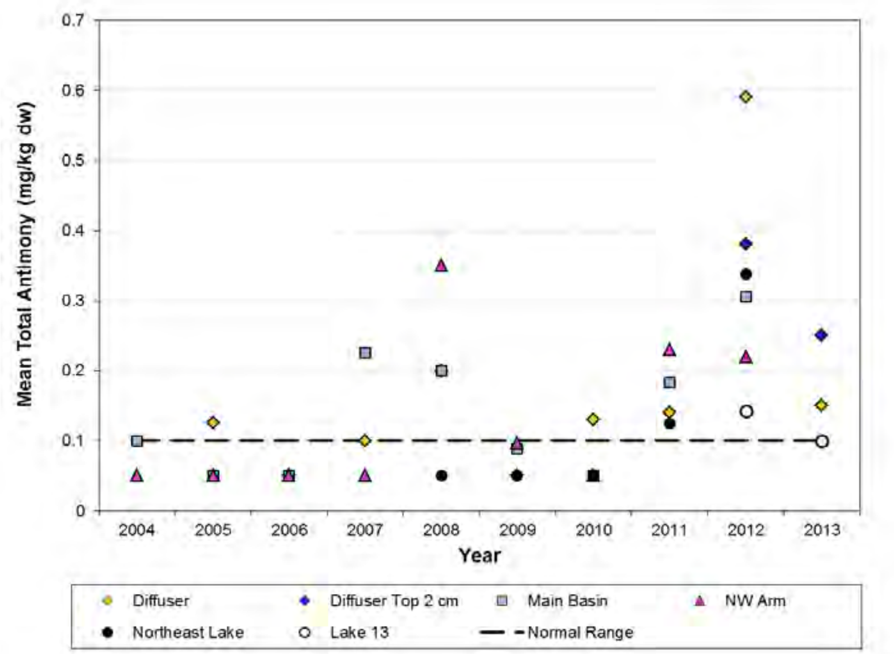
Figure 4-4 Time-Series Plots for Parameters With Statistically Significant Positive Temporal Trends or Concentrations Outside Normal Range at Snap Lake Diffuser Station, 2004 to 2013 (continued)

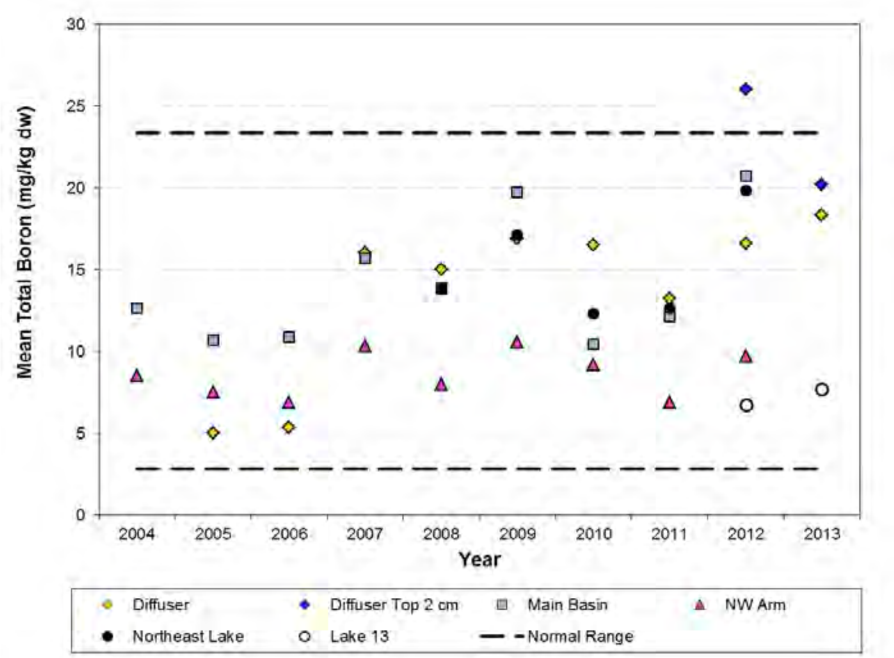




mg/kg dw = milligrams per kilogram dry weight; S = sulphur; cm = centimetre; NW Arm = Northwest Arm.

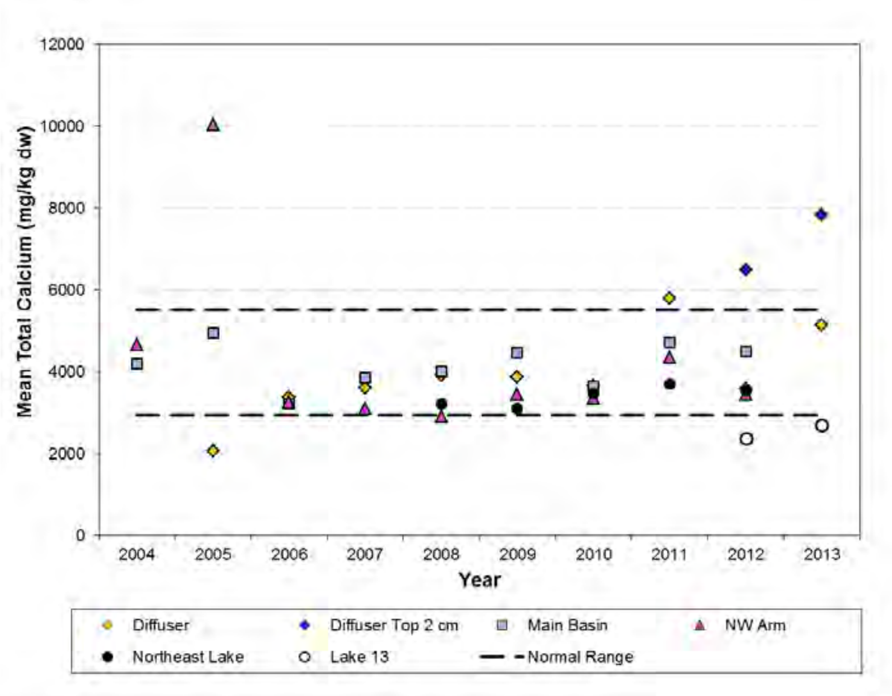
Figure 4-4 Time-Series Plots for Parameters With Statistically Significant Positive Temporal Trends or Concentrations Outside Normal Range at Snap Lake Diffuser Station, 2004 to 2013 (continued)

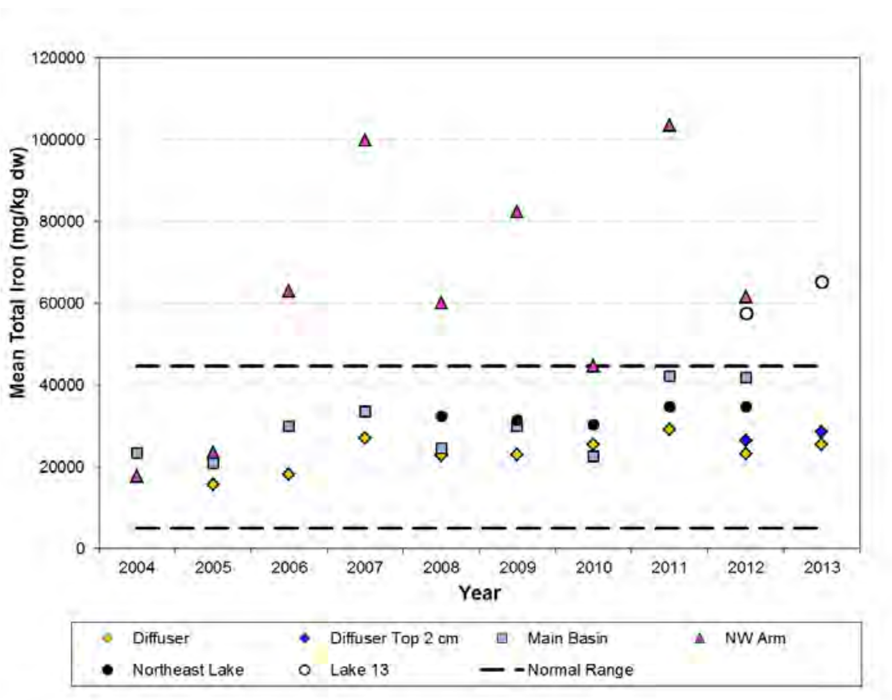




mg/kg dw = milligrams per kilogram dry weight; cm = centimetre; NW Arm = Northwest Arm.

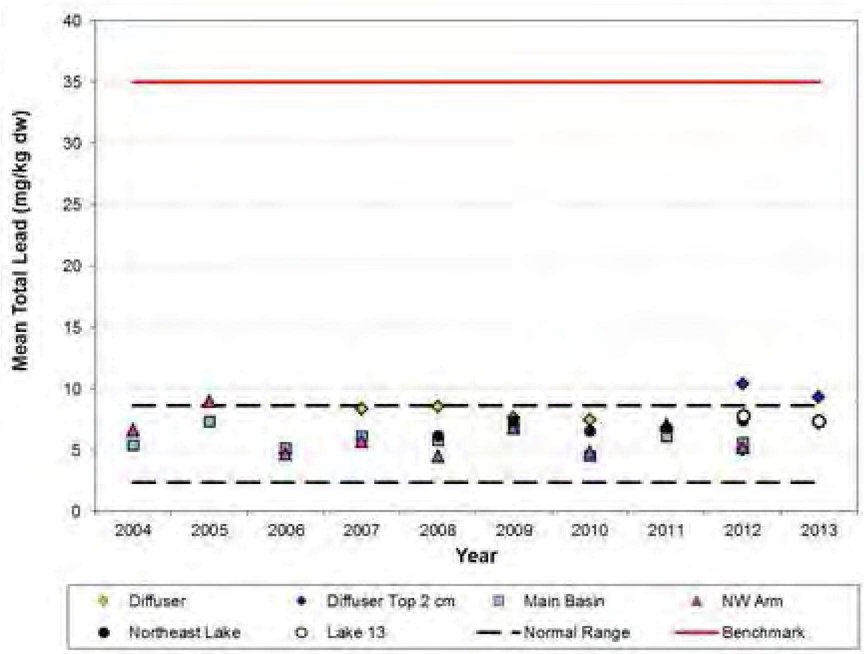
Figure 4-4 Time-Series Plots for Parameters With Statistically Significant Positive Temporal Trends or Concentrations Outside Normal Range at Snap Lake Diffuser Station, 2004 to 2013 (continued)

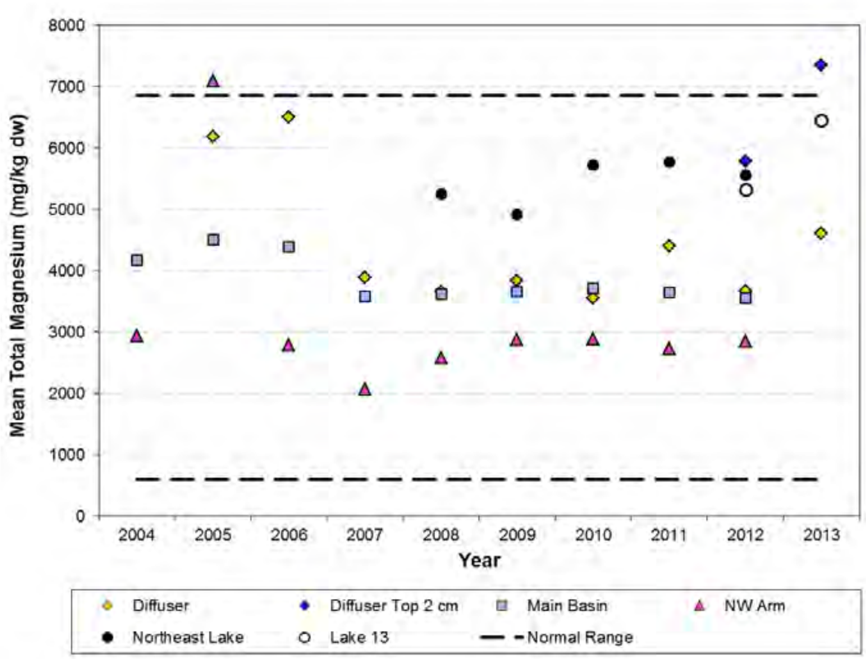




mg/kg dw = milligrams per kilogram dry weight; cm = centimetre; NW Arm = Northwest Arm.

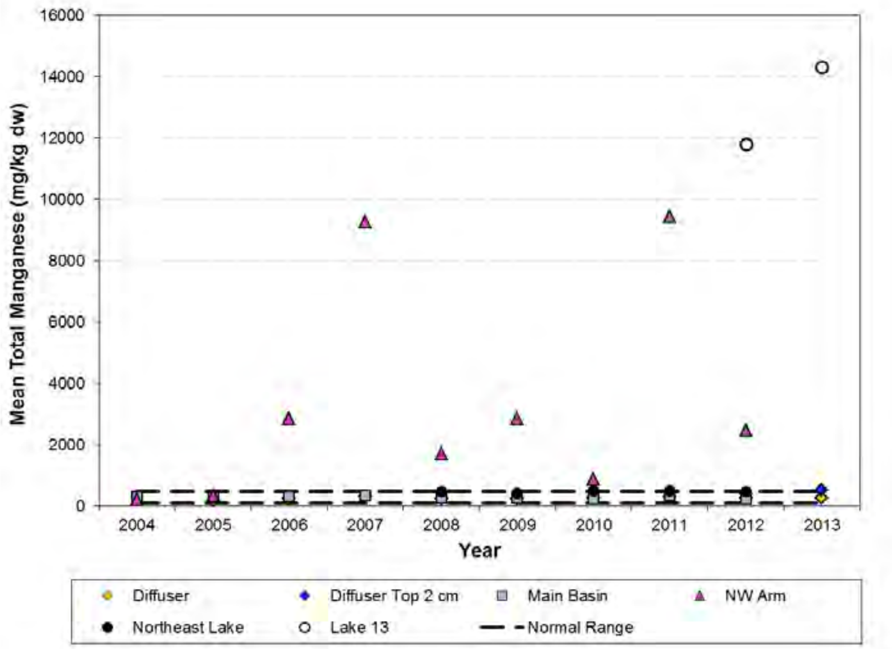
Figure 4-4 Time-Series Plots for Parameters With Statistically Significant Positive Temporal Trends or Concentrations Outside Normal Range at Snap Lake Diffuser Station, 2004 to 2013 (continued)

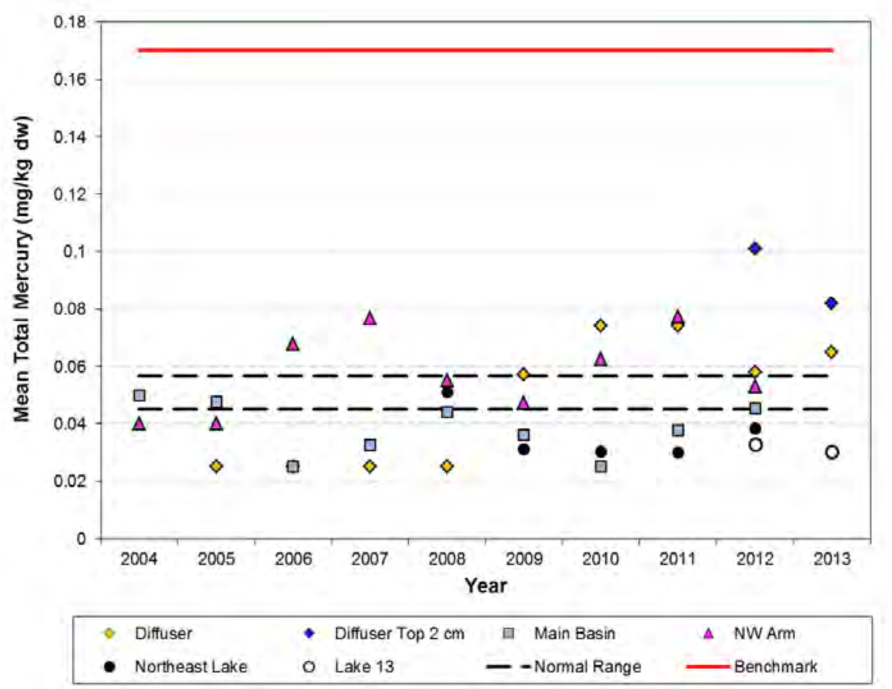




mg/kg dw = milligrams per kilogram dry weight; cm = centimetre; NW Arm = Northwest Arm.

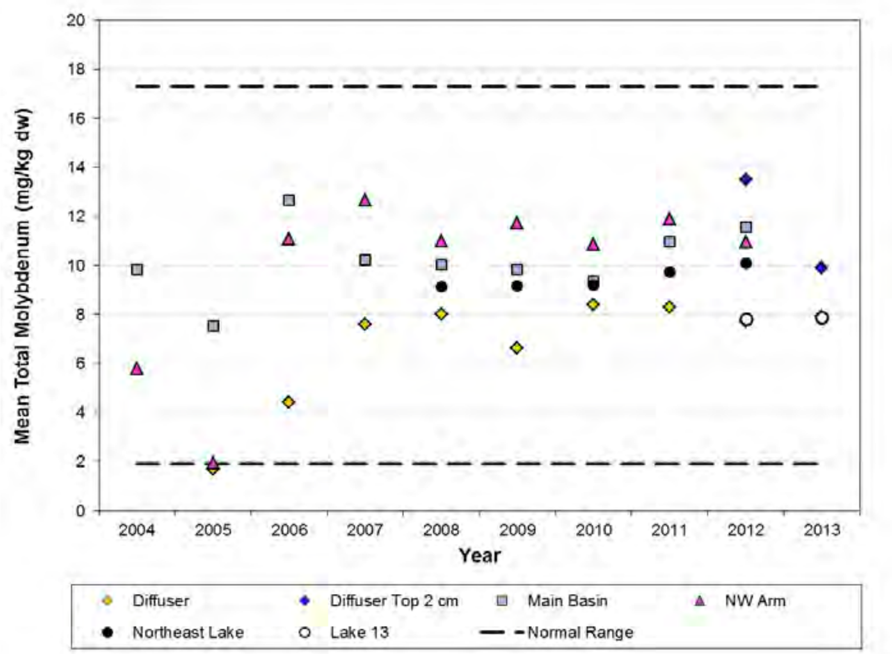
Figure 4-4 Time-Series Plots for Parameters With Statistically Significant Positive Temporal Trends or Concentrations Outside Normal Range at Snap Lake Diffuser Station, 2004 to 2013 (continued)

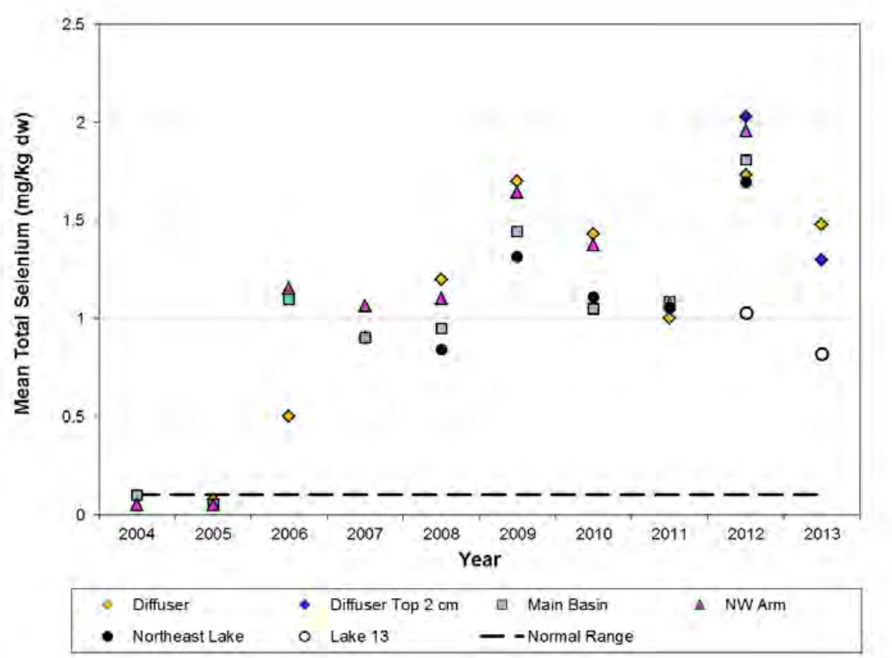




mg/kg dw = milligrams per kilogram dry weight; cm = centimetre; NW Arm = Northwest Arm.

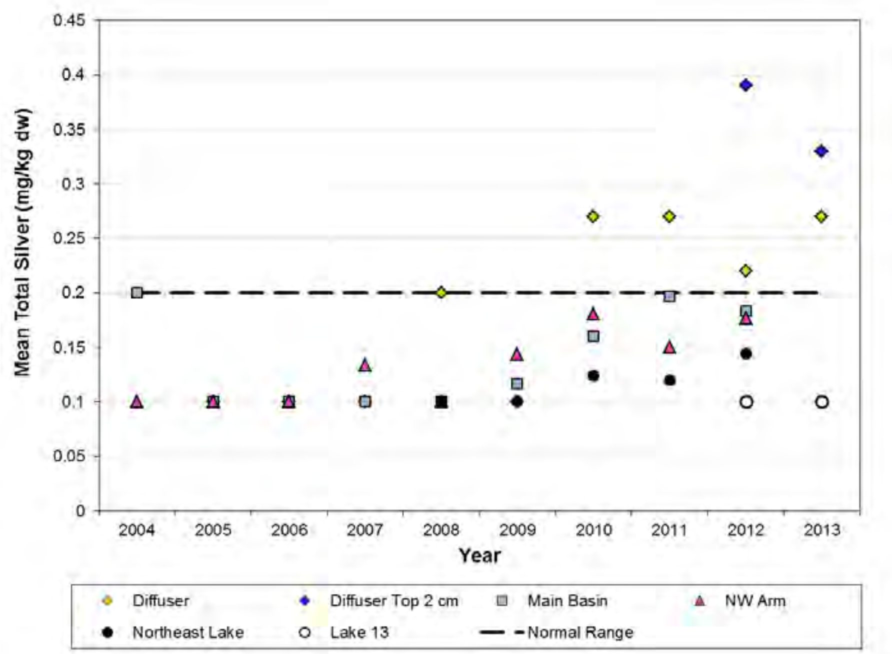
Figure 4-4 Time-Series Plots for Parameters With Statistically Significant Positive Temporal Trends or Concentrations Outside Normal Range at Snap Lake Diffuser Station, 2004 to 2013 (continued)

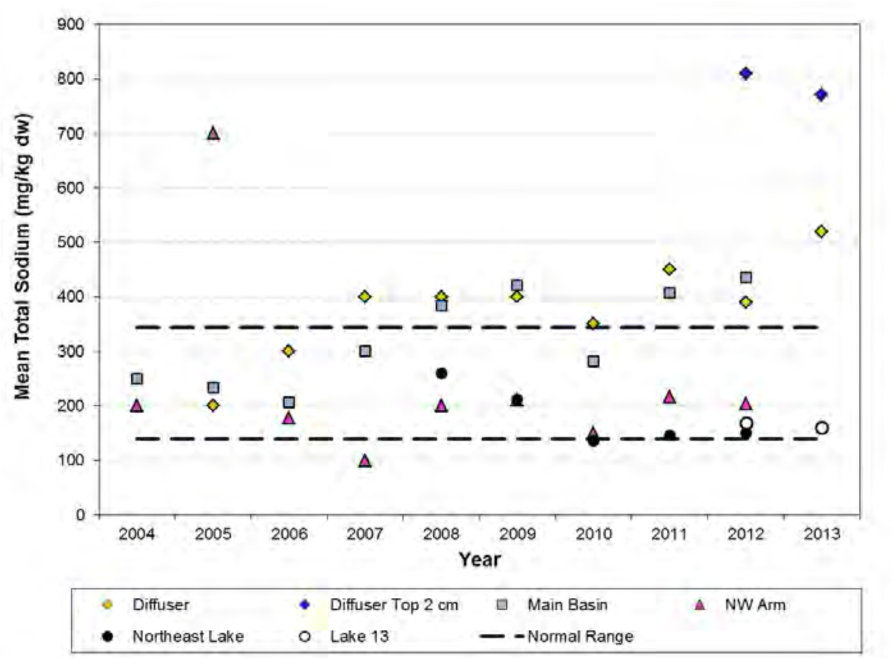




mg/kg dw = milligrams per kilogram dry weight; cm = centimetre; NW Arm = Northwest Arm.

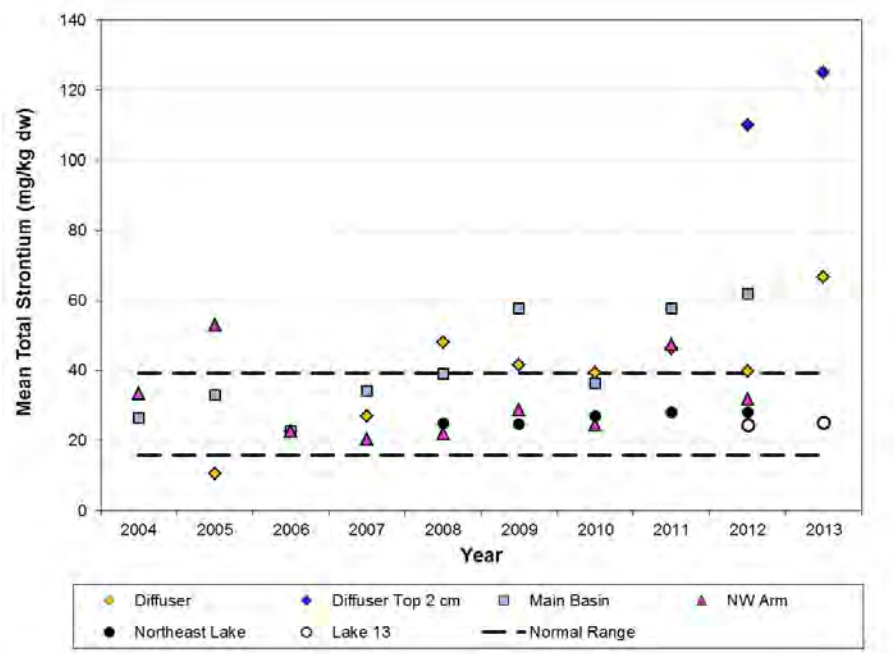
Figure 4-4 Time-Series Plots for Parameters With Statistically Significant Positive Temporal Trends or Concentrations Outside Normal Range at Snap Lake Diffuser Station, 2004 to 2013 (continued)





mg/kg dw = milligrams per kilogram dry weight; cm = centimetre; NW Arm = Northwest Arm.

Figure 4-4 Time-Series Plots for Parameters With Statistically Significant Positive Temporal Trends or Concentrations Outside Normal Range at Snap Lake Diffuser Station, 2004 to 2013 (continued)



mg/kg dw = milligrams per kilogram dry weight; cm = centimetre; NW Arm = Northwest Arm.

Table 4-10 Comparison of 2005 to 2013 Snap Lake Sediment Quality Data to Whole-lake Normal Range

Parameter	Units	Summary Statistics for Calculation of Normal Ranges								Lake Area Mean Concentrations Greater Than Normal Range [a]								
		Year Used	n	Mean	Median	Minimum	Maximum	SD	Normal Range (Mean ± 2SD)	2005 Bulk	2006 Bulk	2007 Top 5-cm	2008 Top 5-cm	2009 Top 5-cm	2010 Top 5-cm	2011 Top 5-cm	2012 Top 5-cm	2013 Top 2 or 5-cm
Physical / Conventional																		
Fines (silt + clay)	% dw	2004	11	93.8	95.0	87.0	97.0	3.2	87.3 - 100.3	-	-	-	-	-	-	-	-	-
Total organic carbon (TOC)	% dw	2004	11	19.5	19.6	7.7	27.3	4.8	9.9 - 29.1	-	-	-	-	-	-	-	-	-
Nutrients																		
Available Ammonium, as N	mg/kg dw	2006	8	50.6	45.9	25.2	71.4	18.3	13.9 - 87.3	-	-	NWA, D, MB	-	-	-	NWA, D, MB	-	-
Available Potassium	mg/kg dw	2005	9	91.8	105.0	41.0	159.0	32.0	27.7 - 156	-	-	-	-	-	-	NWA	-	D
Available Nitrate, as N	mg/kg dw	2005	9	18.8	6.0	1.8	81.0	25.0	0 - 68.8	-	-	-	-	-	-	-	-	-
Available Phosphate, as P	mg/kg dw	2005	9	14.1	8.0	4.0	42.0	12.4	0 - 38.8	-	-	-	D	D	D	D	D	D
Available Sulphate, as S	mg/kg dw	2005	9	75.3	36.0	10.0	234.0	78.9	0 - 233	-	-	-	-	-	-	NWA		D
Total Kjeldahl nitrogen (TKN)	% dw	2004	11	1.44	1.47	0.58	1.93	0.37	0.70 - 2.17	-	-	-	-	-	-	-	-	-
Total nitrogen	% dw	2004	11	1.53	1.55	0.66	1.95	0.34	0.85 - 2.21	-	-	-	-	-	-	-	-	-
Total Metals																		
Aluminum (mg/kg)	mg/kg dw	2004	12	14,933	14,800	8,990	20,300	3,197	8,539 - 21,326	-	-	-	-	-	-	-	-	-
Antimony (mg/kg) [b]	mg/kg dw	2004	12	<0.10	<0.10	<0.10	<0.10	0.00	0.10 - 0.10	D	-	MB	NWA, MB	-	D	NWA, D, MB	NWA, D, MB	D
Arsenic (mg/kg)	mg/kg dw	2004	12	2.83	2.85	1.70	4.40	0.79	1.24 - 4.41	-	-	-	-	NWA	-	-	-	-
Barium (mg/kg)	mg/kg dw	2004	12	215	118	69	1,180	309	0 - 834	-	-	-	-	-	-	-	-	-
Beryllium (mg/kg)	mg/kg dw	2004	12	0.98	1.00	0.60	1.40	0.23	0.51 - 1.44	-	-	-	-	-	-	-	-	-
Bismuth (mg/kg)	mg/kg dw	2004	12	0.53	<0.50	<0.50	0.70	0.06	0.40 - 0.65	-	-	D, MB	D	D	-	D	D, MB	-
Boron (mg/kg)	mg/kg dw	2004	12	13.1	10.5	7.0	22.0	5.1	2.8 - 23.4	-	-	-	-	-	-	-	-	-
Cadmium (mg/kg)	mg/kg dw	2004	12	0.69	0.65	0.50	1.10	0.18	0.34 - 1.05	-	-	-	-	-	-	-	-	-
Calcium (mg/kg)	mg/kg dw	2004	12	4,217	4,000	3,400	5,400	646	2,924 - 5,510	NWA	-	-	-	-	-	-	-	D
Cesium (mg/kg) [c]	mg/kg dw	2004	12	1.88	1.75	1.20	3.90	0.70	0.48 - 3.29	-	-	-	-	-	-	-	-	-
Chromium (mg/kg)	mg/kg dw	2004	12	36.3	35.2	23.9	57.2	9.3	17.6 - 55.0	-	-	-	-	-	-	-	-	-
Cobalt (mg/kg)	mg/kg dw	2004	12	11.6	11.0	8.6	15.9	2.5	6.6 - 16.6	-	NWA	NWA	NWA	NWA	NWA	NWA, MB	NWA	-
Copper (mg/kg)	mg/kg dw	2004	12	99	102	76	118	12	75 - 124	-	-	-	-	-	-	-	-	-
Iron (mg/kg)	mg/kg dw	2004	12	24,650	23,200	9,300	42,100	9,888	4,874 - 44,426	-	NWA	NWA	NWA	NWA	NWA	NWA	NWA	-
Lead (mg/kg)	mg/kg dw	2004	12	5.5	5.2	3.5	9.7	1.6	2.4 - 8.6	D	-	-	-	-	-	-	-	D
Lithium (mg/kg)	mg/kg dw	2004	12	21.0	20.0	13.0	47.0	8.8	3.3 - 38.7	-	-	-	-	-	-	-	-	-
Magnesium (mg/kg)	mg/kg dw	2004	12	3,723	3,470	2,190	8,370	1,566	591 - 6,854	NWA	-	-	-	-	-	-	-	D
Manganese (mg/kg)	mg/kg dw	2004	12	287	279	146	434	96	96 - 478	-	NWA	NWA	NWA	NWA	NWA	NWA	NWA	D
Mercury (mg/kg)	mg/kg dw	2004	12	0.05	<0.05	<0.05	0.06	0.00	0.05 - 0.06	-	NWA	NWA			NWA, D	NWA, D	D	D
Molybdenum (mg/kg)	mg/kg dw	2004	12	9.6	8.9	4.9	18.7	3.9	1.9 - 17.3	-	-	-	-	-	-	-	-	-
Nickel (mg/kg)	mg/kg dw	2004	12	41.6	39.0	34.8	59.4	7.5	26.6 - 56.6	-	-	-	-	-	-	-	-	-
Phosphorus	mg/kg dw	2004	11	1,794	1,820	960	2,750	600	594 - 2,994	-	-	-	-	-	-	-	-	-
Potassium (mg/kg)	mg/kg dw	2004	12	1,742	1,440	990	4,590	954	0 - 3,650	-	-	-	-	-	-	-	-	-
Rubidium (mg/kg)	mg/kg dw	2004	12	13.7	12.0	9.0	33.0	6.5	0.6 - 26.7	-	-	-	-	-	-	-	-	-
Selenium (mg/kg) [b,d]	mg/kg dw	2004	12	<0.10	<0.10	<0.10	<0.10	0.00	0.10 - 0.10	-	NWA, D, MB	NWA, D, MB	NWA, D, MB	NWA, D, MB	NWA, D, MB	NWA, D, MB	NWA, D, MB	D
Silver [b]	mg/kg dw	2004	12	<0.20	<0.20	<0.20	<0.20	0.00	0.20 - 0.20	-	-	-	-	-	D	D	D	D
Sodium (mg/kg)	mg/kg dw	2004	12	242	200	200	300	51	139 - 345	NWA	-	D	D, MB	D, MB	D	D, MB	D, MB	D
Strontium (mg/kg)	mg/kg dw	2004	12	27.4	26.0	21.0	42.0	5.8	15.7 - 39.1	NWA	-	-	-	D, MB	D	NWA, D, MB	D, MB	D
Thallium (mg/kg)	mg/kg dw	2004	12	0.19	0.14	0.07	0.40	0.11	0.00 - 0.41	-	-	-	-	-	-	-	-	-
Tin (mg/kg) [b]	mg/kg dw	2004	12	<2.00	<2.00	<2.00	<2.00	0.00	2.00 - 2.00	-	-	-	-	-	-	-	D, MB	-
Titanium (mg/kg)	mg/kg dw	2004	12	460	436	255	982	181	98 - 822	-	-	-	-	-	-	-	-	-
Uranium (mg/kg)	mg/kg dw	2004	12	9.1	9.6	4.5	13.1	2.8	3.5 - 14.6	-	-	-	-	-	-	-	-	-
Vanadium (mg/kg)	mg/kg dw	2004	12	31.5	30.6	19.3	49.8	7.5	16.6 - 46.4	-	-	-	-	-	-	-	-	-
Zinc (mg/kg)	mg/kg dw	2004	12	185	176	124	321	56	72 - 298	-	-	-	-	-	-	-	-	-

Note: In the "Normal Range" column, lower-range values below zero are shown as zero.
a) Only the diffuser area was assessed in 2013.
b) Concentrations of antimony, selenium, silver, and tin were below their respective detection limits (DLs) at all stations in 2004. The ±2SD ranges for the 2004 lake-wide mean were set equal to their respective DLs.
c) DL for cesium in 2005 was higher (10 mg/kg dw) than that used in all other years; cesium was undetected in all samples in 2005, but the DL was above the ±2SD range for the 2004 lake-wide mean.
d) Selenium was analyzed by the hydride method from 2004 to 2008, by inductively coupled plasma mass spectrometry (ICP-MS) from 2009 to 2012, and by collision cell inductively coupled plasma mass spectrometry (CCMS) in 2013.
± = plus or minus; < = less than; - = mean concentrations were within normal ranges; n = sample size; SD = standard deviation; % dw = percent dry weight; mg/kg dw = milligrams per kilogram dry weight; cm = centimetre; N = nitrogen; P = phosphorus; S = sulphur; NWA = Northwest Arm; D = Diffuser; MB = Main Basin of Snap Lake.

4.4.5 Summary

Snap Lake diffuser station sediments are primarily fine-grained material with elevated TOC concentrations. Sediments from Northeast Lake exhibited similar characteristics to Snap Lake sediments. Although Lake 13 sediments were also primarily fine-grained materials, the TOC concentrations were lower than in Snap Lake and Northeast Lake. Of the 39 nutrient and metal parameters detected in Snap Lake and Lake 13 sediments, 21 parameters had a higher mean concentration in Lake 13 than at the Snap Lake diffuser station.

Concentrations of sediment quality parameters were compared in the top 5-cm and top 2-cm layers at the Snap Lake diffuser station. Differences were considered meaningful if the RPD between the two sampling depths was greater than 20%; a negative RPD meant that the concentration in the top 2-cm layer was greater than the concentration in the top 5-cm layer, a potential indication of Mine-related influence on recently deposited sediments. Percent fines and TOC content were similar at both sampling depths. Fourteen parameters had negative RPDs greater than 20% in 2013: available ammonium; available potassium; available sulphate; antimony; calcium; lead; magnesium; manganese; mercury; molybdenum; nickel; sodium; strontium; and thallium. Apart from available ammonium, available potassium, and thallium, similar results occurred in 2012. Available phosphate was the only parameter to have a positive RPD greater than 20% in both 2012 and 2013, indicating that concentrations were lower in more recently deposited sediments.

Chromium, copper, and zinc concentrations were above their ISQGs at the Snap Lake diffuser station in 2013; concentrations of these metals have been above their ISQGs at this station and elsewhere in Snap Lake in previous years. Arsenic, cadmium, lead, and mercury concentrations have consistently been below their respective ISQGs at the Snap Lake diffuser station since 2005 (except for an anomalous lead measurement in 2005), although concentrations have been above their ISQGs elsewhere in Snap Lake in previous years.

In Lake 13 sediments, concentrations of cadmium, lead, mercury, and zinc were below their respective ISQGs at all five stations, whereas chromium and copper were above their ISQGs for at least four of the five stations in 2013; this was consistent with observations for Snap Lake and Northeast Lake in previous years and suggests that concentrations are naturally elevated in sediments in the area surrounding the Mine. Arsenic concentrations were higher in Lake 13 sediments than at the Snap Lake diffuser station, exceeding the ISQG at all stations and the PEL at one station. One station (LK13-03) in the northeast area of Lake 13 had consistently higher concentrations of arsenic and several other metals in both 2012 and 2013.

Temporal trends were evaluated for the Snap Lake diffuser area, using data from top 5-cm layer and bulk samples collected between 2005 and 2013. Results showed that 27 parameters showed increasing trends and 12 showed decreasing trends, but these were not all statistically significant. Twelve parameters had statistically significant increasing trends: available potassium; available sulphate; aluminum; boron; calcium; iron; mercury; molybdenum; selenium; silver; sodium; and, strontium. Four parameters had statistically significant decreasing trends at the Snap Lake diffuser station: barium; cesium; thallium; and, titanium.

Comparison of sediment parameter concentrations at the Snap Lake diffuser station with their baseline normal ranges showed that concentrations of available potassium, available phosphate, available sulphate, antimony, calcium, lead, magnesium, manganese, mercury, selenium, silver, sodium, and strontium were above their respective normal ranges in 2013. Antimony, selenium, and silver were not detected in baseline sediment samples; therefore, their normal ranges are equal to their respective DLs and any detected concentrations will be above the normal range.

Overall sediment quality at the Snap Lake diffuser station, and the potential for Mine-related effects, was assessed through comparison to SQGs, comparison of concentrations at two sampling depths, analysis for temporal trends, and comparison to baseline normal ranges. The integrated results of these comparisons showed that concentrations of available sulphate, calcium, mercury, sodium, and strontium were potentially being influenced by Mine operations, as reflected in higher concentrations in recently deposited sediments, increasing concentrations over time, and exceedence of normal ranges. Mercury was the only parameter with a CCME SQG; mercury has only been detected in Snap Lake sediments in recent years and always below its ISQG, and is therefore not expected to adversely affect biota.

4.5 Conclusions

The three key questions listed below are intended to apply to assessment of sediment quality in the main basin of Snap Lake, which is now performed every three years. For 2013, they were assessed with respect to conditions at the Snap Lake diffuser station.

4.5.1 Are Concentrations of Sediment Quality Parameters Above or Below Sediment Quality Guidelines?

Exceedances of the SQGs available for seven metals were documented. Concentrations of chromium, copper, and zinc were above their respective ISQGs at one or both Snap Lake diffuser station samples in 2013; similar results were reported in previous years at this station but ISQG exceedances did not occur every year. Concentrations of arsenic, chromium, and copper were above their respective ISQGs in Lake 13 samples; arsenic was above the PEL at one station.

4.5.2 Are there Differences in Sediment Quality in Snap Lake Relative to the Reference Lake and, if so, are they Related to the Mine?

Differences in sediment quality in Snap Lake relative to the reference lakes are intended to be assessed by making statistical comparisons of mean parameter concentrations in the Snap Lake main basin with mean concentrations in Northeast Lake and/or Lake 13. For 2013, data were only available for the Snap Lake diffuser station and Lake 13, as per the 2013 AEMP Design Plan. Of the 39 nutrient and metal parameters detected in sediment samples in 2013, 21 parameters had higher mean concentrations in Lake 13 than at the Snap Lake diffuser station.

Comparison of sediment parameter concentrations in the top 5-cm and top 2-cm layers at the Snap Lake diffuser stations showed that 11 parameters had higher concentrations (defined as having a negative RPD greater than 20%) in the thinner sediment layer in both 2012 and 2013: available sulphate; antimony; calcium; lead; magnesium; manganese; mercury; molybdenum; nickel; sodium; and, strontium. This indicates the potential for Mine-related increases in these parameter concentrations.

4.5.3 Are Concentrations of Sediment Quality Parameters Increasing over Time?

Temporal trends in sediment parameter concentrations were only assessed for the Snap Lake diffuser station in 2013. Twelve parameters showed statistically significant increasing trends over the period from 2005 to 2013: available potassium; available sulphate; aluminum; boron; calcium; iron; mercury; molybdenum; selenium; silver; sodium; and, strontium. Four parameters had statistically significant decreasing trends at the Snap Lake diffuser station: barium; cesium; thallium; and, titanium.

4.6 Recommendations

Recommendations for modifications to the sediment quality component of the Snap Lake AEMP are identified below.

- Continue to use Northeast Lake and Lake 13 as reference lakes to assess long-term regional trends, but exclude the anomalous LK13-03 station from calculation of mean parameter concentrations for Lake 13.

4.7 References

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SECTION 5

PLANKTON

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LIST OF ACRONYMS

Term	Definition
AEMP	Aquatic Effects Monitoring Program
Bio-Limno	Bio-Limno Research and Consulting Inc.
$CaCO_3$	calcium carbonate
De Beers	De Beers Canada Inc.
DO	dissolved oxygen
EAR	Environmental Assessment Report
Golder	Golder Associates Ltd.
i.e.	that is
LK13	Lake 13
MVLWB	Mackenzie Valley Land and Water Board
Mine	Snap Lake Mine
N	nitrogen
N_2	atmospheric nitrogen
NEL	Northeast Lake
PAR	photosynthetically active radiation
P	phosphorus
PI	prediction interval
QA	quality assurance
QC	quality control
RPD	relative percent difference
SD	standard deviation
SE	standard errror
Si	silica
SNAP	Snap Lake
TN	total nitrogen
TP	total phosphorus
TDS	total dissolved solids
TSS	total suspended solids
UofA	University of Alberta Biogeochemical Analytical Laboratory
WOE	weight of evidence
X	times

UNITS OF MEASURE

Term	Definition
%	percent
% SI	percentage of surface irradiance
±	plus or minus
<	less than
>	greater than
°C	degrees Celsius
cells/L	cells per litre
K*z*	attenuation coefficient
L	litre
m	metre
mg/L	milligrams per litre
mg/m^3	milligrams per cubic metre
mL	millilitre
mm	millimetre
m/s	metres per second
mm^3/m^3	cubic millimetres per cubic metre
org/m^3	organisms per cubic metre
µmol photons/s/m^2	micromoles of photons per second per square metre
µg/L	micrograms per litre
µm	micrometre
µS/cm^2	microSiemens per square centimetre
W/m^2	watts per square metre

5 PLANKTON

5.1 Introduction

De Beers Canada Inc. (De Beers) owns and operates the Snap Lake Mine (Mine) northeast of Yellowknife, in the Northwest Territories. The Mine operated under the terms and conditions of a Class A Water Licence since 2004 (Licence #MV2001L2-0002; MVLWB 2004). On June 14, 2012, the Water Licence was renewed for another eight year period (Licence #MV2011L2-0004; MVLWB 2013).

The Aquatic Effects Monitoring Program (AEMP) is a requirement of the Water Licence, Part G (MVLWB 2013). The goal of the AEMP is to address potential Mine-related effects to the aquatic ecosystem of Snap Lake in a scientifically defensible and cost-effective manner.

The assessment endpoints for the AEMP are based on the valued ecosystem components identified in the Environmental Assessment Report (EAR), the effect predictions in the EAR, and narrative commitments made by De Beers during the EAR process (De Beers 2002) and through the Environmental Agreement (De Beers 2004). De Beers committed that water quality, fish health, and ecological function will remain acceptable in Snap Lake. The plankton monitoring component of the AEMP addresses the ecological function assessment endpoint.

Overall, two main impact hypotheses are examined in the AEMP for the Mine; both of these impact hypotheses are addressed in the plankton component of the AEMP:

- *Toxicological Impairment Hypothesis*: Toxicity to aquatic organisms could occur due to substances of toxicological concern (primarily metals and total dissolved solids [TDS]) released to Snap Lake; and,
- *Nutrient Enrichment Hypothesis*: Eutrophication could occur due to the release of nutrients (primarily phosphorus [P] and nitrogen [N], and, for some species, TDS) to Snap Lake.

5.1.1 Background

5.1.1.1 Phytoplankton

The term “plankton” is a general term referring to small, usually microscopic organisms that live suspended in the water. For the purpose of this study, the term “phytoplankton” refers to the algal component of the plankton community, ranging between 2 and 20 micrometres (μm) in size. Phytoplankton can be grouped into the following eight major groups:

- Cyanobacteria;
- Chlorophyceae (chlorophytes);
- Chrysophyceae (chrysophytes);

- Cryptophyceae (cryptophytes);
- Bacillariophyceae (diatoms);
- Dinophyceae (dinoflagellates);
- Euglenophyceae (euglenoids); and,
- Xanthophyceae (xanthophytes).

A full taxonomic analysis of the phytoplankton community provides the best estimate of biomass through biovolume measurements and also provides useful taxonomic information that can be used to assess community changes. Understanding community changes is important, as excess limiting nutrients can encourage the growth of certain phytoplankton groups, such as cyanobacteria, which may produce harmful toxins. While additional nutrients can change plankton communities by adding biomass and/or changing community composition, other substances can be toxic and can change plankton communities by reducing biomass while still changing community composition.

Phytoplankton pigments, such as chlorophyll *a, b,* and *c* can be used to understand algal viability and the health of the phytoplankton community. Algal viability within the community is important as it can be a major driver of primary production (Franklin et al. 2012). Chlorophyll *a* is the primary photosynthetic pigment contained in phytoplankton, which is why it is widely used as a surrogate measure of phytoplankton biomass. However, chlorophyll *a* concentrations can be affected by changes in environmental conditions, such as light, nutrient availability, and temperature, as well as by phytoplankton community composition (Healey 1975).

In 2011, as part of the AEMP Annual Report (De Beers 2012a), the relationship between chlorophyll *a* and total phytoplankton biomass in Snap Lake was evaluated and found to be poor. It was recommended that chlorophyll *a* should not be used as a surrogate for total phytoplankton biomass in Snap Lake; however, monitoring of this parameter was continued as required by the Water License MV2001L2-0002 (MVLWB 2004), and the renewed Water Licence MV2011L2-0004 (MVLWB 2013).

5.1.1.2 Zooplankton

The term “zooplankton” refers to microscopic animals and includes Rotifera (rotifers) and crustaceans, specifically Cladocera (cladocerans), Cyclopoida (cyclopoid copepods), and Calanoida (calanoid copepods). Cyclopoid and calanoid copepods are considered separately because of taxonomic and ecological differences. Calanoids are typically herbivorous, feeding on phytoplankton while cyclopoids are typically omnivorous, feeding on phytoplankton and small zooplankton (Brönmark and Hansson 1998). Additionally, calanoids are almost exclusively pelagic (i.e., open-water), while cyclopoids are dominated by littoral (i.e., near-shore) species, although a few pelagic species of cyclopoids can account for a major component of the planktonic community.

5.1.1.3 Influence of Nutrients

The primary nutrients limiting the development of phytoplankton in nature are P, N, and for diatoms, silica (Si). Planktonic community structure, composition, and biomass can be better understood by examining the distribution, supply, and composition of these nutrients in Snap Lake.

Elevated concentrations of P and N can lead to large increases in phytoplankton biomass (i.e., blooms). These blooms can occur throughout the water column and can prevent light from reaching the waters below. This stops the growth of plants and other algae found deeper in the water column and reduces biological diversity. In addition, when phytoplankton die, they settle out of the water column and undergo bacterial decomposition, which uses up oxygen in the water and reduces oxygen availability for other biota (Wetzel 2001).

Changes in Si concentration can also result in changes in phytoplankton community composition. In particular, diatoms use inorganic Si to create biogenic Si for their cell walls. In lakes with long residence times, Si can be depleted by diatom growth and subsequent sinking of their frustules (their hard and porous cell walls) to the sediments, resulting in a selective advantage for algal groups that do not require Si for growth. Silica is considered limiting for diatom growth at concentrations below 100 micrograms per litre (µg/L; Reynolds 2006).

5.1.1.4 Influence of Potentially Toxic Substances

Several mine-related stressors can be toxic to plankton and can contribute to shifts in community structure as well as increased mortality. Changes in the ionic composition of TDS can cause exclusions of some zooplankton species while promoting population growth in others (Derry et al. 2003), whereas TDS has no predictable relationship with phytoplankton biomass (Prepas 1983). Both zooplankton and phytoplankton are susceptible to changes in lake pH, i.e., acidification can alter dissolved metals, such as copper, cadmium, and aluminum, into toxic reactive species. Specifically, these changes can inhibit P uptake in phytoplankton such as *Scenedesmus quadricauda* (Peterson et al. 1984) and can change the chemical composition within the body of zooplankton, such as *Daphnia middendorffiana*, increasing their susceptibility to pathogen infection (Havas 1985). Metals including aluminum, cadmium, copper, lead, mercury, nickel, and zinc can be toxic to *Daphnia magna* (Attar and Maly 1982; Havas 1985; Khangarot and Ray 1989). Reactive metals such as cadmium can also affect phytoplankton communities leading to inhibitive effects on cyanobacateria photosynthesis (Zhou et al. 2006).

5.1.1.5 Influence of Light

Solar radiation, or sunlight, consists of different wavelengths of light. Photosynthetically active radiation (PAR) is the specific band of solar radiation that is used by plants for photosynthesis. The intensity of solar radiation within the water column influences aquatic life, such as phytoplankton, littoral algae, and macrophytes that rely on light for photosynthesis and growth. The euphotic zone is where photosynthesis occurs; it extends from the surface of the water to a depth where PAR is approximately one percent (%) of light measured at the surface (i.e., PAR greater than 1% of subsurface incident radiation). Once light enters the water, light intensity decreases logarithmically with water depth (Wetzel 2001). This is quantified by the extinction or attenuation coefficient, which is the fraction of light that is absorbed per meter and is related to reflection, refraction, or scattering, and absorption by water, dissolved compounds, and suspended particles (Dodds and Whiles 2010). The maximum depth of the euphotic zone depends on the extent of this light attenuation in the water column.

Light data are critical to defining the extent of the euphotic habitats occupied by phytoplankton. Light can be measured by Secchi depth (i.e., the depth that the Secchi disk disappears and then reappears from view). Secchi depth corresponds to the depth at which approximately 10% of the surface light remains (Dodds and Whiles 2010); however, there are inherent sources of variance in Secchi depth measurements that make it a coarse and semi-quantitative method of measuring PAR (Preisendorfer 1986). The relationship between transmission of PAR and Secchi depth is complex and nonlinear because it depends on ambient light, scattering, and absorptive properties of the water. When measuring light to understand biological processes, such as photosynthesis, it is important to measure total light coming from all directions including above and below, which cannot be achieved with a Secchi disk. Furthermore, Secchi depth readings are highly dependent on the visual acuteness of the observer at the moment of measurement (Preisendorfer 1986).

A more direct, quantitative and less subjective measure of PAR can be obtained with a light meter. Light meter data can also be used to assess the potential impact of increased total suspended solids on water transparency over time. Typically in more productive (eutrophic) waterbodies with large phytoplankton biomass or with large amounts of suspended or dissolved materials, the water contains more material to absorb or reflect light, which inhibits light transmittance to deeper depths. Conversely, in less productive (oligotrophic) waterbodies with low amounts of suspended or dissolved material, light is transmitted to greater depths (Dodds and Whiles 2010).

5.1.1.6 Food Web Functionality: Phytoplankton Edibility

Not all ecosystems function with the same efficiency. One characteristic of “healthy” ecosystems is efficient transport of metabolically available energy and critical nutrients through the base of the food web. One of the major characteristics of “unhealthy” plankton ecosystems is inefficient transfer of primary production from phytoplankton to zooplankton (Makarewicz 1993). Both nutrient enrichment and toxicological impairment can lead to altered phytoplankton community composition and ultimately to food web inefficiency due to diminished carbon-transfer through the grazing food chain.

The availability of growth-limiting nutrients (i.e., N, P, and Si) and changes in potentially toxic substances (i.e., dissolved metals and TDS) can lead to alterations in phytoplankton community composition, which in turn may alter the efficiency of grazing by zooplankton. Some phytoplankton taxa can efficiently use low concentrations of available nutrients, achieving a competitive advantage. Such “oligotrophic" species succeed because of their superior ability to utilize critical nutrients at low concentrations (Sommer 1989). Generally, these taxa are considered edible and can be efficiently grazed by zooplankton.

Other phytoplankton taxa can utilize nutrients contributed by enrichment for rapid growth; these taxa can displace species with slower growth rates and high affinity for limiting nutrients.

At low nutrient concentrations, phytoplankton with the highest affinity for limiting nutrients, a high surface area to volume ratio, and tolerance to potentially toxic substances will become dominant. At high nutrient input rates, an increase in larger phytoplankton species (diatoms), which are not under the direct numerical control of their grazers, generally dominate. With continued nutrient inputs, a niche is created for larger, poorly edible, or inedible phytoplankton species (Anderson et al. 2002).

Resistance against grazing can be accomplished through sufficient size, chemical and mechanical defense, toxicity, and a combination of these factors (Turner and Tester 1997). Many of the cyanobacteria species are capable of rapid growth under nutrient sufficiency and are considered inedible taxa that are poorly grazed by zooplankton. The occurrence and persistence of toxin-forming cyanobacteria blooms are other factors that result in resistance to grazing.

In response to nutrient enrichment, zooplankton grazing rates can vary, which can influence phytoplankton species composition. Increased phytoplankton biomass can translate into increased food availability for zooplankton and, ultimately, fish. However, fish and predatory zooplankton species have the potential to alter phytoplankton community structure (Carpenter and Kitchell 1984; Lampert et al. 1986; McQueen and Post 1988). The size of the zooplankton and their grazing pressure can select for larger species of phytoplankton. *Daphnia* spp. are large cladoceran zooplankton capable of intense grazing and may be responsible for triggering seasonal blooms of larger colonial inedible phytoplankton species, such as species of cyanobacteria and chlorophytes (Lampert et al. 1986). Cladoceran abundances in Snap Lake historically have been low and the zooplankton community in Snap Lake is copepod-dominated (De Beers 2012b). Grazing rates of copepods are lower than cladocerans; as a result, copepods do not have as great of an effect on the phytoplankton community structure as cladocerans (Wetzel 2001).

In general, low zooplankton grazing rates would favour edible phytoplankton species. High zooplankton grazing rates would favour inedible species because nutrients regenerated from the digestion of edible species provide a continuous nutrient supply for rapid growth of inedible taxa (Sterner 1989). With abundant available nutrients, inedible, grazer-resistant phytoplankton species are capable of forming harmful blooms that can cause negative effects ranging from toxin production and associated drinking water concerns to nuisance blooms. Nuisance blooms have a high biological oxygen demand during decomposition and can lead to depleted oxygen concentrations in deep waters, causing detrimental effects on higher trophic organisms such as fish (Anderson et al. 2002).

5.1.1.7 Plankton as a Monitoring Tool

Phytoplankton and zooplankton community metrics can be useful indicators of environmental change because of their rapid response to changes in nutrients or other substances. However, the inherent variability within the plankton community poses a challenge and also limits their usefulness as a monitoring tool. Plankton abundance, biomass, and taxonomic composition vary vertically and horizontally within the open-water; therefore, estimates are sensitive to the number of stations, samples, and the depth of the water column sampled (Findlay and Kling 2001; Paterson 2002). Seasonal succession within the plankton community and natural year-to-year variation also contribute to the inherent variability of these communities (Wetzel 2001; Paterson 2002). As a result, intensive sampling of plankton is required to collect sufficient data for meaningful interpretation of the monitoring results.

5.1.2 Objective

The principal objective of the 2013 plankton (phytoplankton and zooplankton) monitoring component of the AEMP for the Snap Lake Mine was to meet the following Water Licence MV2011L2-0004 (Part G, Schedule 6, Item 1) (MVLWB 2013) conditions:

a) Monitoring for the purpose of measuring Project-related effects on the following components of the Receiving Environment:

viii. the communities of zooplankton and phytoplankton due to changes in water quality;

b) Monitoring the following as indicators of nutrient enrichment in Snap Lake:

ii. chlorophyll a and algal biomass and species composition of the phytoplankton community.

c) Monitoring to verify or assess the Environmental Assessment predictions relating to the trophic and dissolved oxygen status of Snap Lake including monitoring of:

iv. Concentrations of chlorophyll a in Snap Lake in early summer after the loss of ice cover and mid-summer; and,

v. Algal biomass and species composition for phytoplankton in Snap Lake in mid-summer. The monitoring should include measures of cyanobacteria biomass and species composition and cyanotoxins in the event that algal community compositions shift to favour cyanobacteria.

Analyses of the 2013 plankton component data addressed the following key questions:

- What are the current concentrations of chlorophyll *a* and *c* and what do these concentrations indicate about the trophic status of Snap Lake, Northeast Lake, and Lake 13?
- What is the current status, in terms of abundance, biomass, and composition, of the phytoplankton community in Snap Lake, Northeast Lake, and Lake 13 and do these results suggest Mine-related nutrient enrichment or toxicological impairment?

- What is the current status, in terms of abundance, biomass, and composition, of the zooplankton community in Snap Lake, Northeast Lake, and Lake 13 and do these results suggest Mine-related nutrient enrichment or toxicological impairment?
- How do observed changes compare to applicable predictions in the EAR?

5.2 Methods

5.2.1 Sampling Locations

In 2013 plankton sampling stations were consistent with stations monitored under the water quality component, following the 2013 AEMP Design Plan (De Beers 2014). Sampling occurred at nine monitoring stations in Snap Lake, with five stations in the main basin (SNAP02-20e, SNAP03, SNAP06, SNAP08, and SNAP11A) and four stations in the northwest arm (SNAP02A, SNAP20B, SNAP23, and SNAP29; Figure 5-1). Two reference lakes were also sampled. Five monitoring stations were sampled in Northeast Lake (NEL01, NEL02, NEL03, NEL04, and NEL05; Figure 5-2), and five monitoring stations were sampled in Lake 13 (LK13-01, LK13-02, LK13-03, LK13-04, and LK13-05; Figure 5-3).

5.2.2 Timing of Sampling

To accurately assess seasonal variability of the plankton communities in Snap Lake, Northeast Lake, and Lake 13, sampling occurred monthly during the open-water period between July and September. A summary of the sampling events completed in Snap Lake, Northeast Lake, and Lake 13 is presented in Table 5-1.

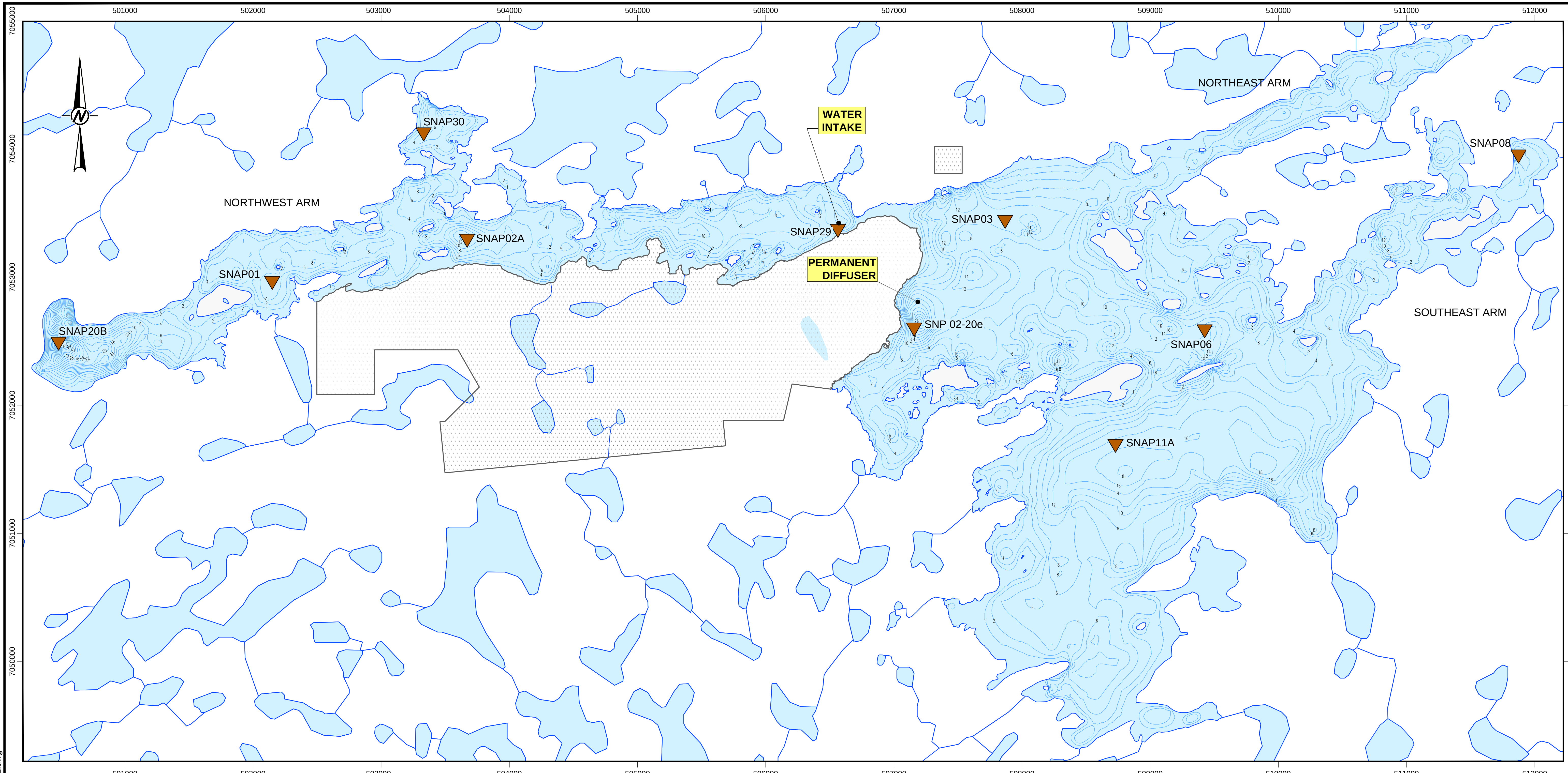
Table 5-1 Summary of Plankton Community Sampling Events in Snap Lake, Northeast Lake, and Lake 13, 2013

Variable	July 7 to 16, 2013 (n)	August 8 to 14, 2013 (n)	September 5 to 15, 2013 (n)
Main Basin of Snap Lake			
Secchi Depth	5	5	5
LI-COR Light Reading	5	5	5
Phytoplankton	5	5	5
Chlorophyll *a* and *c*	10	10	10
Zooplankton	5	5	5
Northwest Arm of Snap Lake			
Secchi Depth	**4**	**4**	**4**
LI-COR Light Reading	4	4	0
Phytoplankton	4	4	5
Chlorophyll *a* and *c*	8	8	8
Zooplankton	4	4	5

Table 5-1 Summary of Plankton Community Sampling Events in Snap Lake, Northeast Lake, and Lake 13, 2013

Variable	July 7 to 16, 2013 (n)	August 8 to 14, 2013 (n)	September 5 to 15, 2013 (n)
Northeast Lake			
Secchi Depth	0	5	5
LI-COR Light Reading	5	5	5
Phytoplankton	5	5	5
Chlorophyll *a* and *c*	10	10	10
Zooplankton	5	5	5
Lake 13			
Secchi Depth	4	5	5
LI-COR Light Reading	5	5	5
Phytoplankton	5	5	5
Chlorophyll *a* and *c*	10	10	10
Zooplankton	5	5	5

n = number of samples.



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LEGEND

- Depth Contour (m)
- Watercourse
- Waterbody
- Snap Lake Mine Footprint
- Plankton Monitoring Station

REFERENCES

Digital Map from Mackay lake, Northwest Territories, produced by Department of Energy, Mines and Resources. Map 75M, Original scale 1:250,000, projection : Transverse Mercator, Datum : NAD83, coordinate system : UTM zone 12.

Snap Lake outline and islands were corrected to LANDSAT 7 satellite image 45/15, dated September 2, 2000. provided by Geobase.

Bathymetry was created in Surfer 8 using sonar data from the 2002 North Lakes program (Golder) and 2005 transect data from the Reference Lake Search program (Golder).

NOTES

1. Sample collection included plankton monitoring component and picoplankton special study.

SNAP LAKE MINE

Plankton Monitoring Stations in Snap Lake, 2013

PROJECTION: UTM Zone 12	DATUM: NAD83	DE BEERS GROUP OF COMPANIES
500 0 500 SCALE 1:30,000 METRES		
FILE No: 1313490001-7010_5-1		DATE: February 21, 2014
JOB No: 14-1349-0004	REVISION No: A	
OFFICE: GOLDER VICTORIA	DRAWN: JEF CHECK: PMC	Figure 5-1

LEGEND

- Depth Contour (m)
- Watercourse
- Waterbody
- Plankton Monitoring Station

REFERENCES

Digital Map from Mackay lake, Northwest Territories, produced by Department of Energy, Mines and Resources. Map 75M, Original scale 1:250,000, projection : Transverse Mercator, Datum : NAD83, coordinate system : UTM zone 12.

Lake outline and islands were corrected to LANDSAT 7 satellite image 45/15, dated September 2, 2000. provided by Geobase.

Bathymetry was created in Surfer 8 using sonar data from the 2002 North Lakes program (Golder) and 2005 transect data from the Reference Lake Search program (Golder).

NOTES

1. Grid is displayed in Transverse Mercator, Datum : NAD83, Coordinate system : UTM zone 12.
2. Sample collection included plankton monitoring component and picoplankton special study.

SNAP LAKE MINE

Plankton Monitoring Stations in Northeast Lake, 2013

PROJECTION: UTM Zone 12	DATUM: NAD83	DE BEERS GROUP OF COMPANIES
SCALE 500 0 500 METRES		
FILE No: 1313490001-7010_5-2		DATE: February 21, 2014
JOB No: 14-1349-0004	REVISION No: A	Figure 5-2
OFFICE: GOLDER VICTORIA	DRAWN: JEF	CHECK: PMC

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LEGEND

- Depth Contour (m)
- Watercourse
- Waterbody
- Plankton Monitoring Station

REFERENCES

Digital Map from Mackay lake, Northwest Territories, produced by Department of Energy, Mines and Resources. Map 75M, Original scale 1:250,000, projection : Transverse Mercator, Datum : NAD83, coordinate system : UTM zone 12.

Bathymetry was created in Surfer 8 using sonar data from the 2002 North Lakes program (Golder) and 2005 transect data from the Reference Lake Search program (Golder).

NOTES

1. Grid is displayed in Transverse Mercator, Datum : NAD83, Coordinate system : UTM zone 12.
2. Sample collection included plankton monitoring component and picoplankton special study.
3. Station LK13-01 was sampled in 2012 for chlorophyll a, plankton monitoring component and picoplankton special study.

SNAP LAKE MINE

Plankton Monitoring Stations in Lake 13, 2013

PROJECTION: UTM Zone 12	DATUM: NAD83
SCALE 500 0 500 METRES	DE BEERS GROUP OF COMPANIES
FILE No: 1313490001-7010_5-3	DATE: Febraury 21, 2014
JOB No: 14-1349-0004	REVISION No: A
OFFICE: GOLDER VICTORIA	DRAWN: JEF
CHECK: PMC	Figure 5-3

5.2.3 Sampling Methods

5.2.3.1 Supporting Environmental Variables

Depth profiles of pH, dissolved oxygen (DO), conductivity, and water temperature, consisting of measurements at the water surface and continuing to the lake bottom, were measured at each station during each sampling event in Snap Lake, Northeast Lake, and Lake 13. Detailed methods are presented in Section 3.0. Secchi depths were also recorded at each sampling station during each sampling event.

5.2.3.2 Light Assessment

Light profiles were measured using a LI-COR LI-1400 light meter with a spherical light sensor to simultaneously measure upwelling (light being reflected back from below the sensor) and downwelling (light entering the water from above the sensor). The LI-COR light meter measured PAR as micromoles of photons per second per square metre (μmol photons/s/m^2). The LI-COR light meter was lowered into the water and light measurements were recorded at the surface (considered 100% irradiance), at 0.25 metres (m), 0.5 m, and 1.0 m, and then every subsequent metre until the LI-COR light meter reached the bottom of the water column (Appendix 5A). Light profiles were not measured during the September field program at SNAP20B, SNAP02A, SNAP23, SNAP29, and SNAP06 due to equipment malfunction. The field crews were able to collect light profile readings at all remaining stations in Snap Lake and both reference lakes.

All light measurements were expressed as a percentage of the surface irradiance value (% SI) calculated as follows:

$$\% \text{ SI} = (I_z/I_0) \times 100 \qquad \text{[Equation 5-1]}$$

where I_z and I_0 are irradiance (μmol photons/s/m^2) at depth z (metres) and at the surface, respectively.

The vertical light attenuation coefficient (K*z*) was also calculated to compare light attenuation through the water column at different stations and on different sampling occasions. Light attenuation was calculated using the transformed Beer-Lambert equation as follows:

$$Kz = -[\ln(I_z/I_0)]/z \qquad \text{[Equation 5-2]}$$

where *Kz* is the attenuation coefficient at a specific depth z (metres) and
I_z and I_0 are irradiance (μmol photons/s/m^2) at depth z and at the surface, respectively.

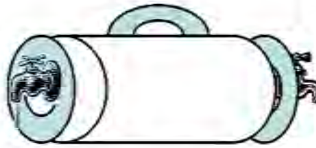
5.2.3.3 Phytoplankton

Sample collection methods for the phytoplankton sub-component are summarized in Figure 5-4. In Snap Lake, Northeast Lake, and Lake 13, the upper 6 m of the water column were sampled at all stations, with the exception of SNAP08, where the water depth was less than 6 m. The top 6 m of the water column is the estimated euphotic zone in these lakes, where light is sufficient for phytoplankton photosynthesis. Water was collected using a 2-litre (L) Kemmerer water sampler at 2-m intervals (i.e., 0 m, 2 m, 4 m, and 6 m) to the maximum depth of the estimated euphotic zone. Equal volumes of water from each depth were combined in a clean 11-L bucket and then transferred into appropriate sample containers for the following composite samples:

- chlorophyll *a* and *c* (presented in the current section);
- microcystin-LR (presented in Section 3.0);
- total nitrogen (TN) and total phosphorus (TP; presented in Section 3.0);
- phytoplankton taxonomy (presented in the current section); and,
- picoplankton and flagellate taxonomy (presented in Section 11.2).

Chlorophyll *a* and *c*: Two composite chlorophyll samples were collected from each station resulting in 18 chlorophyll samples per sampling event in Snap Lake, 10 chlorophyll samples per sampling event in Northeast Lake, and 10 chlorophyll samples in Lake 13 (Table 5-1). Chlorophyll samples were collected in amber Nalgene bottles to prevent degradation from exposure to light. For each chlorophyll sample, 250 or 500 millilitres (mL) of water were filtered through a 47 millimetre (mm) diameter Whatman GF/C glass fibre filter; the volume filtered for each sample was recorded, and the data were corrected to account for differences in the volume filtered. The chlorophyll filtration was done under low light conditions in the laboratory. The process was repeated resulting in two samples per station. The filters were frozen and shipped to the University of Alberta Biogeochemical Analytical Laboratory (U of A), in Edmonton, Alberta, where chlorophyll *a* and *c* analyses were completed.

Phytoplankton Taxonomy: A single composite phytoplankton sample was collected at each station, resulting in nine samples per sampling event in Snap Lake, five samples per sampling event in Northeast Lake, and five samples per sampling event in Lake 13. Phytoplankton samples were collected in amber Nalgene bottles to prevent degradation from exposure to light. Samples were preserved with 10 mL of Dafano’s and 2.5 mL of Lugol’s solutions and kept at room temperature. Phytoplankton samples and supporting information (i.e., sample depth and volume) were sent to Bio-Limno Research and Consulting Inc. (Bio-Limno), in Halifax, Nova Scotia, for analyses of taxonomic composition, abundance, and biomass.



Water collected from 0, 2, 4, and 6 m (when possible) and combined to create a composite water sample; samples distributed as follows:

Chlorophyll *a* and *c*
Collect 2 L of the composite water in an amber plastic bottle and return to laboaratory to filter for two chlorophyll *a* and *c* samples.

Phytoplankton Taxonomy
Collect one 250 mL sample from the composite water in amber plastic bottles

Picoplankton Special Study [a]
Collect two 250 mL samples from the composite water in amer plastic bottles, one for picoplankton enumeration and one for flagellate taxonomy.

Place individual filters into petri dishes, wrap in foil and freeze.

Phytoplankton and flagellate samples preserved with acidic Lugol's solution; picoplankton sample preserved with 10% buffered formalin.

Collect a single vertical zooplankton tow from 1 m off the lake bottom into 250 mL clear plastic bottle.

Place one half of an Alka-Seltzer tablet into the sample; allow to dissolve and then preserve with 10% buffered formalin.

NOTES

(a) Sampling related to the Picoplankton Special Study will continue as part of the plankton component.

SNAP LAKE MINE		
Overview of the Plankton Sample Collection Methods		
PROJECTION: N/A	DATUM: N/A	DE BEERS GROUP OF COMPANIES
SCHEMATIC ONLY, NOT TO SCALE		
FILE No: 1313490001-7010_5-4		DATE: February 25, 2014
JOB No: 14-1349-0004	REVISION No: A	Figure **5-4**
OFFICE: GOLDER VICTORIA	DRAWN: JEF / CHECK: PMC	

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5.2.3.4 Zooplankton

A single zooplankton sample was collected at each station, resulting in nine samples per sampling event in Snap Lake, five samples per sampling event in Northeast Lake, and five samples per sampling event in Lake 13. Zooplankton samples were collected using a 0.30 m diameter, 153 µm Nitex mesh plankton net with a detachable Dolphin bucket. A Rigo flow meter (model number 5571-A) was attached to the mouth of the plankton net. The flow meter readings (number of revolutions) were recorded and used to verify towing consistency in the field. The plankton net was lowered to 1 m off the lake bottom and then towed vertically to the water surface. The sample was then concentrated in the Dolphin bucket and transferred to a 250-mL white Nalgene bottle.

Inconsistency was noted in the haul rate for the plankton net during all three sampling periods. The July sampling period had consistently low revolutions in metres per second (m/s) whereas, during the September sampling period, haul rates were high. The August haul rates were consistent among stations. In September, the flow meter did not give a reading at NEL01.

In the field, each zooplankton sample was treated with one-half of an Alka-Seltzer tablet, which was added to prevent the zooplankton from contorting, which makes taxonomic identification difficult. Each sample was then preserved with 10% buffered formalin. Sample depths were recorded for use by the taxonomist for calculating abundance and biomass on a volumetric basis. Samples were kept at room temperature and sent to Bio-Limno for analyses of taxonomic composition, abundance, and biomass.

5.2.4 Sample Sorting and Taxonomic Identification

Phytoplankton and zooplankton taxonomy samples were analyzed by Bio-Limno according to the methods provided below.

Phytoplankton: Aliquots of the preserved phytoplankton samples were allowed to settle overnight in sedimentation chambers following the procedure of Lund et al. (1958). Algal units were counted from randomly selected transects on a Zeiss Axiovert 40 CFL inverted microscope. Counting units were individual cells, filaments, or colonies depending on the organization of the phytoplankton. A minimum of 400 units were counted for each sample. The majority of the samples were analyzed at 500 times (X) magnification, with initial scanning for large and rare organisms (e.g., *Ceratium* sp.) completed at 250 X magnification. Taxonomic identifications were based primarily on: Geitler (1932); Skuja (1949); Huber-Pestalozzi (1961, 1972, 1982, 1983); Findlay and Kling (1976); Anton and Duthie (1981); Prescott (1982); Whitford and Schumacher (1984); Starmach (1985); Tikkanen (1986); Krammer and Lange-Bertalot (1986, 1988, 1991a,b); Komárek and Anagnostidis (1998, 2005); and, Wehr and Sheath (2003).

Wet weight biomass was calculated from recorded abundance and specific biovolume estimates based on geometric solids (Rott 1981), assuming unit specific gravity. The biovolume, in units of cubic millimetres per cubic metre (mm^3/m^3) wet weight of each species, was estimated from the average dimensions of 10 to 15 individuals. The biovolumes of colonial taxa were based on the number of individuals within each colony. All calculations for cell concentration and biomass were performed with Hamilton's (1990) computer program.

Zooplankton: Three 1 to 5 mL sub-samples were removed from each sample for identification and enumeration of zooplankton taxa. Exact volumes of each sub-sample were dependent on the amount of particulate material present in the sample. Macro-zooplankton consisting of cladocerans, cyclopoids, and calanoids, were identified and enumerated using a dissecting microscope at magnifications between 12 and 50 X. An inverted microscope at magnifications between 200 and 400 X was used to identify and enumerate rotifers and copepod nauplii. Sub-samples for rotifers and nauplii were allowed to settle for 24 hours before counting. Taxonomic identifications were based primarily on the following: Brooks (1957); Edmondson (1966); Chengalath et al. (1971); Grothe and Grothe (1977); Pennak (1978); Stemberger (1979); Clifford (1991); and, Thorp and Covich (1991).

Zooplankton lengths were determined directly on the microscope fitted with a micrometre inside the ocular lens. In general, lengths were measured on a minimum of 40 to 50 individuals of each species or genus encountered within a representative subset of samples. Length measurements were recorded for rare taxa or those that occurred in low numbers as they were encountered. Wet weight biomass was calculated for each sample, based on published length-weight regressions cited in Botterell et al. (1976), Downing and Rigler (1984), and Stemberger and Gilbert (1987). For each sample, mean individual weights for each species were calculated by averaging estimated weights. Total biomass for each species or developmental stage was calculated as the product of its abundance and estimated mean individual weight.

5.2.5 Data Analyses

5.2.5.1 Approach

The plankton component analyses were designed to answer the Key Questions listed in Section 5.1.2; an overview of the analytical approach associated with each key question is provided in Table 5-2. The annual evaluation of the plankton component was focused on qualitative assessments of spatial, seasonal, and annual trends in the main basin of Snap Lake, northwest arm of Snap Lake, Northeast Lake, and Lake 13. In addition, quantitative statistical assessments were performed to examine relationships between plankton biological variables and select environmental variables. Calculations were completed using Microsoft Office Excel 2007. Scatter plots were created in Sigma Plot Version 12.0 (SYSTAT 2011) and statistical analyses were performed in SYSTAT 13 (SYSTAT 2009). Specific details relevant to the data analyses methods to address each key question are provided in Sections 5.2.6.2 to 5.2.6.5.

Table 5-2 Overview of Analysis Approach for Plankton Component Key Questions

Key Question	Overview of Analysis Approach
1. What are the current concentrations of chlorophyll *a* and *c* and what do these concentrations indicate about the trophic status of Snap Lake, Northeast Lake, and Lake 13?	Temporal trends in chlorophyll *a* and *c* concentrations were examined and current concentrations were compared to trophic classifications outlined in the EAR (De Beers 2002).
2. What is the current status, in terms of abundance, biomass and composition, of the phytoplankton community in Snap Lake, Northeast Lake, and Lake 13 and do these results suggest signs of Mine-related nutrient enrichment or toxicological impairment?	Qualitative comparisons were completed, comparing the current Snap Lake phytoplankton community to the reference lakes and to baseline (i.e., 2004). Supporting information from other components was used to assess any habitat-related variation. Changes in the proportion of edible and inedible phytoplankton were examined using spatial and temporal trend analyses.
3. What is the current status, in terms of abundance, biomass and composition, of the zooplankton community in Snap Lake, Northeast Lake, and Lake 13 and do these results suggest signs of Mine-related nutrient enrichment or toxicological impairment?	Qualitative comparisons were completed, comparing the current Snap Lake zooplankton community to the reference lakes and to baseline (i.e., 2004). Supporting information from other components was used to assess any habitat-related variation.
4. How do observed changes compare to applicable predictions in the EAR?	A qualitative assessment of annual trends in Snap Lake was completed and compared to the EAR predictions.

EAR = Environmental Assessment Report.

5.2.5.2 Key Question 1: What are the current concentrations of chlorophyll a and c and what do these concentrations indicate about the trophic status of Snap Lake, Northeast Lake, and Lake 13?

Spatial and temporal trends in chlorophyll *a* and *c* concentrations were examined and current concentrations in Snap Lake, Northeast Lake, and Lake 13 were compared to trophic classifications outlined in the EAR.

5.2.5.3 Key Question 2: What is the current status, in terms of abundance, biomass and composition, of the phytoplankton community in Snap Lake, Northeast Lake, and Lake 13 and do these results suggest signs of Mine-related nutrient enrichment or toxicological impairment?

A qualitative review of the phytoplankton data was performed, in the form of spatial and temporal trend analyses. Trend analyses compared mean (plus or minus [±] standard error [SE]) annual Snap Lake phytoplankton communities to the reference lakes and to baseline (i.e., 2004). This information, in combination with the weight of evidence (WOE) assessment (Section 7.0), was used to determine whether abundance, biomass, or community composition in Snap Lake show signs of Mine-related nutrient enrichment or toxicological impairment.

Phytoplankton abundance and biomass data were divided into groups based on taxonomic results as follows: cyanobacteria; chlorophytes; chrysophytes; cryptophytes; dinoflagellates; diatoms; and, others. The relative proportion accounted for by each group, based on both abundance and biomass, was calculated separately for each station and for each sampling event to evaluate temporal and spatial (lake-to-lake) variability in community structure. Seasonal and spatial (within lake) variability in 2013 was assessed by examining total biomass and the biomasses of each major group.

Information collected as part of the water quality component (maximum water depth, Secchi depth, conductivity, DO, water temperature, pH, and nutrient concentrations) and the supporting environmental variables component (cloud cover, solar radiation, air temperature, water levels, and annual temperature logger data) were incorporated to assess habitat-related changes in the phytoplankton community. Spearman rank order correlations were used to assess potential habitat-related variation. Nutrient data including P, N, and Si concentrations from the water quality component were compared to phytoplankton community data to assess potential Mine-related eutrophication. Toxicity data from the water quality component were assessed and compared to plankton community data to assess the potential for Mine-related toxicological impairment.

In addition, an annual edibility assessment was completed on the phytoplankton data as described in Section 5.2.7. Changes in the proportion of edible and inedible phytoplankton (based on size and potential toxicity) were visually examined using spatial and temporal trend analyses. These changes were then related to changes in zooplankton abundance and biomass to gain a better understanding of any trophic effects on zooplankton.

5.2.5.4 Key Question 3: What is the current status, in terms of abundance, biomass and composition, of the zooplankton community in Snap Lake, Northeast Lake, and Lake 13 and do these results suggest signs of Mine-related nutrient enrichment or toxicological impairment?

A qualitative review of the zooplankton data was performed, in the form of spatial and temporal trend analyses. Trend analyses compared mean (± SE) annual Snap Lake zooplankton communities to the reference lakes and to baseline (i.e., 2004). This information, in combination with the WOE assessment (Section 7.0), was used to determine whether abundance, biomass, or community composition in Snap Lake show signs of Mine-related nutrient enrichment or toxicological impairment.

Zooplankton abundance and biomass data were divided into groups based on taxonomic results as follows: cladocerans; calanoid copepods; cyclopoid copepods; and, rotifers. The relative proportion accounted for by each group, based on both abundance and biomass, was calculated separately for each station and for each sampling event to evaluate temporal and spatial (lake-to-lake) variability in community structure. Seasonal and spatial (within lake) variability in 2013 was assessed by examining total biomass and the biomass of each major group.

Information collected as part of the water quality component (maximum water depth, Secchi depth, conductivity, DO, water temperature, pH, and nutrient concentrations) and the supporting environmental variables component (cloud cover, solar radiation, air temperature, water levels, and annual temperature logger data) were incorporated to assess habitat-related changes in the zooplankton community. Spearman rank order correlations were used to assess potential habitat-related variation. Nutrient data including P, N, and Si concentrations from the water quality component were compared to phytoplankton community data to assess potential Mine-related eutrophication. Toxicity data from the water quality component were assessed and compared to plankton community data to assess the potential for Mine-related toxicological impairment.

5.2.5.5 Key Question 4: How do observed changes compare to applicable predictions in the EAR?

A qualitative assessment of annual trends in Snap Lake was completed and compared to the following EAR predictions:

- a gradual increase in chlorophyll *a* from 0.2 to 1.8 μg/L to 1.5 to 2.3 μg/L, with chlorophyll *a* levels remaining within the range associated with oligotrophic lakes and no change in the overall trophic status of Snap Lake;
- a slight increase in algal abundance and biomass and, to a lesser extent, zooplankton abundance and biomass, leading to a minor increase in fish food;
- minor changes in phytoplankton and zooplankton community structure, with a potential change in the relative proportion of various species, but no major shifts in keystone species and no loss of species; and,
- a gradual lake-wide increase in TDS concentration, which would lead to an increase in calcium concentrations (to 110 mg/L) in Snap Lake, the effects of which would be negligible on phytoplankton, but have low magnitude effects on zooplankton, specifically cladocerans.

5.2.6 Edibility Assessment

Phytoplankton taxa identified in Snap Lake, Northeast Lake, and Lake 13 between 2004 and 2013 were separated into two categories: edible; and, inedible. In the absence of size data for each taxon, the phytoplankton taxa were classified as edible or inedible based on rationale provided in Table 5-3. There is a level of uncertainty in the classification of phytoplankton taxa as edible or inedible, as the size class of each identified taxon was not provided by the taxonomist (De Beers 2012b).

The relative proportions of edible and inedible biomass were calculated for each open-water period by year in the main basin and northwest arm of Snap Lake, Northeast Lake, and Lake 13. In addition, total edible phytoplankton biomass and inedible phytoplankton biomass were plotted against zooplankton biomass in Snap Lake and Northeast Lake to assess potential grazer-induced changes in the phytoplankton community; Lake 13 was not included in this assessment due to insufficient data, sufficient data will be available for the 2016 re-evaluation.

Table 5-3 Rationale for Classification of Phytoplankton Taxa as Edible or Inedible

Major Group	Edible	Inedible
Cyanobacteria	n/a	• all taxa classified as inedible • may produce toxins that can have a negative effect on plankton • cells are protected by a mucilaginous sheath • filamentous taxa can attain a length that inhibits the filter feeding apparatus of zooplankton
Chlorophytes	• unicellular taxa • flagellates	• colonial taxa (with or without mucilaginous sheath) • filamentous taxa that can attain a length that inhibits the filter feeding apparatus of zooplankton
Chrysophytes	• all taxa classified as edible	n/a
Cryptophytes	• all taxa classified as edible	n/a
Dinoflagellates	• athecate (i.e., lacking a hard shell) taxa and taxa with maximum size <40 µm	• thecate taxa (i.e., possessing a hard shell that makes ingestion difficult for zooplankton) • maximum size >40 µm
Diatoms	• unicellular • maximum size <40 µm	• stellate or ribbon colonies • maximum size >40 µm

n/a = not applicable; < = less than; > = greater than; µm = micrometre.

5.2.7 AEMP Response Framework Action Levels for Plankton

The Response Framework for the Snap Lake AEMP provides a systematic approach to responding to the results of the AEMP such that the potential for significant adverse effects is identified and any necessary mitigation actions are undertaken. This process is described in Section 13. The two core values of concern (valued ecosystem components) that were identified in the EAR were drinking water and fish. Within these two core values, Significance Thresholds were determined. The Significance Threshold associated with the plankton component is related to ecological function, specifically as inadequate food for fish (i.e., a persistent decline in total phytoplankton abundance or biomass beyond the level of natural variability, or a persistent absence of cladocerans from Snap Lake).

Evaluation of Action Levels related to biological responses is difficult as ecosystem components, such as plankton, can exhibit toxicity or enrichment responses. In addition, plankton communities are inherently variable; therefore, persistent changes need to be observed before action is taken. Changes are considered ecologically important or persistent if they are maintained for three or more years. The time-frame of three years is necessary given the naturally high variability in the plankton community as reflected in AEMP monitoring results to date (De Beers 2012b).

A change is documented if differences are observed between Snap Lake and the reference lakes, or if current indicators of change are outside the normal range. The normal range could not be calculated using the reference lake data because differences in the trophic status of Snap Lake and Northeast Lake limited these direct comparisons (De Beers 2012b). In addition there are no established benchmark values or critical effect sizes for plankton endpoints; thus, comparisons of the magnitude of effect to a single value, i.e., ± two standard deviations (SDs), cannot be used to assess ecological significance for plankton. The normal range can also not be based on a single year's data, i.e., it cannot be based on the 2004 baseline data because there is no measure of year-to-year variability. Therefore, the normal range for plankton was based on prediction interval (PI) equations as outlined in Appendix 5B. The data were $\log_{10}$ transformed to obtain a bell-shaped distribution for the normal range calculations; all subsequent data analyses were conducted using $\log_{10}$ transformed data, and Action Level analyses are based on this $\log_{10}$ transformed data.

For the Toxicological Impairment Action Levels for plankton, total phytoplankton biomass was selected as the overall indicator for the phytoplankton community, and cladoceran abundance and biomass were selected for the zooplankton community (Table 5-4). For the Nutrient Enrichment Action Levels, total phytoplankton and zooplankton community structure were selected as the overall indicators of phytoplankton and zooplankton community function (Table 5-4). A shift in community structure is also an indicator of change and at the "major" group level of phytoplankton or zooplankton community composition is considered important. The "major" group-level refers to the Class level of biological organization for phytoplankton and the Phylum (i.e., Rotifera) or Order level (i.e., Cladocera, Calanoida, Cyclopoida) of biological organization for zooplankton. Changes at the species or genus-level occur regularly from year to year within the plankton community; therefore, examining the community at a higher level of biological organization is required. In addition, for the Nutrient Enrichment Action Levels, total phytoplankton biomass and total zooplankton biomass were selected as indicators for the phytoplankton and zooplankton communities, respectively. Biomass is clearly an important endpoint relative to predation. Some level of nutrient enrichment is expected and, at low levels, nutrient enrichment may be beneficial to the plankton community. Thus, a more persistent effect on the plankton community is required to reach the Low Action Level for nutrient enrichment, compared to the Low Action Level for toxicological impairment.

Table 5-4 Action Levels for Plankton

Tiered Action Level	Toxicological	Nutrient Enrichment
Key Information	Differences between Snap Lake and reference lakes or normal range	Differences between Snap Lake and reference lakes or normal range
Negligible	No persistent decline beyond the normal range in total phytoplankton biomass or cladoceran abundance and biomass	No consistent ecologically-important changes in richness and community structure
Low	A persistent decline beyond the normal range in total phytoplankton biomass within the main basin of Snap Lake **OR** A persistent decline beyond the normal range in cladoceran abundance **or** biomass within the main basin of Snap Lake	Persistent increase beyond the normal range in total phytoplankton or zooplankton biomass in the main basin of Snap Lake **AND** Minor shift in phytoplankton or zooplankton community composition (based on major[(a)] groups) in the main basin of Snap Lake
Medium	*TBD*[(b)]	*TBD*[(b)]
High	*TBD*[(b)]	*TBD*[(b)]

Note: "Normal Range" is determined based on the prediction interval as outlined in Appendix 5B.

a) "Major" indicates a change at the Class level of biological organization for phytoplankton and a combination of Phylum and Order levels for zooplankton.

b) TBD = to be determined if Low Action Level is reached.

5.2.8 Weight-of-Evidence Assessment

The WOE approach described in Section 7.0 is applied to integrate the annual AEMP results to assist in understanding the underlying cause(s) of biological responses to stressors. Whereas the Response Framework assesses each component separately to determine whether changes in individual AEMP components are sufficiently significant to warrant response actions, the WOE approach integrates measures of exposure (e.g., water quality, sediment quality) with measures of biological response (e.g., plankton, benthos, fish), in a given year, to assess the underlying causes of biological changes, i.e., nutrient enrichment or toxicological impairment. Thus, the WOE approach links biological effects to exposure with the purpose of supporting Response Planning if and when warranted by the Action Levels.

The analyses described in the preceding sections feed into the WOE approach described in Section 7.0. The WOE approach applies a rating scheme to determine the degree of change in AEMP components and then proceeds with integrating related components.

5.3 Quality Assurance and Quality Control

5.3.1 Overview of Procedures

Quality assurance (QA) and quality control (QC) procedures were applied during all aspects of the plankton component to check that the data collected were of acceptable quality. Data entered electronically were reviewed for data errors.

Ten percent of both the phytoplankton and zooplankton samples were re-counted by the same taxonomist to verify counting efficiency. The inherent variability associated with the plankton samples prevents the establishment of a QC threshold value. For the purposes of the plankton QC, the proportion of each major group was calculated and the occurrence of dominant species used to assess consistency between the field samples and duplicate samples analyzed. In addition, the Bray-Curtis index and relative percent difference (RPD) were used to assess the overall similarity between the field and duplicate samples. Due to high variability in species occurrence, the comparisons were made at the major group level for both abundance and biomass rather than the species level. The Bray-Curtis index only allows for comparison between entire samples, while the RPD compares differences in abundance and biomass of each major group between each pair of duplicate samples. Values that had RPDs greater than 50% were flagged and follow-up assessments of the data were performed.

The Bray-Curtis index is a measure of ecological distance between two communities and is used to assess the overall similarity between the taxonomist's original and recounted samples. All values greater than 0.5 were flagged and follow-up discussions with the taxonomist were initiated. The value of the Bray Curtis index ranges from 0 (identical communities) to 1 (very dissimilar communities) and is calculated using the formula:

$$b = \frac{\sum_{k=1}^{n} \left| x_{ik} - x_{jk} \right|}{\sum_{k=1}^{n} \left(x_{ik} + x_{jk} \right)}$$

[Equation 5-3]

where xik and xjk are abundance from the original and re-counted samples respectively.

Flagged data were not automatically rejected because of exceedance of the acceptance criterion; rather, they were evaluated on a case-by-case basis, as some level of within-station variability is expected for taxonomy samples. If there were departures from the acceptance criterion, a variety of follow-up assessments were performed. These assessments included plotting the data for visual identification of outliers. If there were visual outliers, the data were plotted with the corresponding 2004 to 2012 data for a range comparison. If the data were outside the corresponding 2004 to 2012 range, laboratory re-analysis occurred. If laboratory re-analysis confirmed the results, the outlier points were retained in the final data set unless there was a technically defensible reason to exclude them.

The data were also reviewed for unusually high or low values (i.e., greater or less than 10 times typical lake values), which would suggest erroneous results. Unusually high or low results were invalidated on a case-by-case basis. Invalidated data were retained in Appendix 5C tables, but a flag of "XC" was appended to the data, indicating that the sample was considered unreliable or the results were designated as not correct due to an internal review of the data.

5.3.2 Summary of Quality Assurance and Quality Control Results

The 2013 Snap Lake AEMP plankton QC procedure outlined in the design document specified that 10% of both the phytoplankton and zooplankton samples would be re-counted by a third-party taxonomist, to verify the counting efficiency of the primary taxonomic analyses (De Beers 2014). However, although the same detailed methods were provided to the primary and third-party taxonomists, the methods utilized by the third-party taxonomist differed from the methods used by the primary taxonomist. In addition to differences caused by variation between taxonomists, it was determined that the data generated using the original plankton QC design may have been unreliable due to the large sample volume from which each subsample originated. A re-evaluation of the Golder Associates Ltd. (Golder) plankton QC procedures is necessary before a similar comparison is attempted in the next annual AEMP report.

Further Golder QC procedures were employed in 2013 within the primary taxonomy lab. Specifically, 10% of both the phytoplankton and zooplankton samples were re-counted by the same taxonomist, to verify internal counting efficiency. A total of six phytoplankton and six zooplankton samples were re-processed for QC analyses.

The phytoplankton results were consistent between the original field samples and primary taxonomist's QC samples. Total phytoplankton abundance compared between sub-samples had RPD values less than 50%, and Bray-Curtis values less than 0.5 for all phytoplankton QC samples (Appendix 5C, Table 5C-1). For phytoplankton biomass, QC analyses indicated that subsamples from three of the six QC samples (SNAP08, SNAP11A, and LK13-03) had significant differences between the original sample and the taxonomist QC sample in the total phytoplankton biomass estimate (i.e., RPDs greater than 50% and Bray-Curtis values greater than 0.5; Appendix 5C, Table 5C-2); however, due to the level of consistency, the phytoplankton data is considered reliable.

The zooplankton QC analyses indicated that most results were consistent between the original sample and the primary taxonomist's QC sample, indicating good reliability for zooplankton data. Total zooplankton abundance compared between QC subsamples had RPD values less than 50%, and Bray-Curtis values less than 0.5 (Appendix 5C, Table 5C-3). For zooplankton biomass, QC analyses indicated that subsamples from two of the six QC samples (NEL03 and NEL05) had significant differences in total zooplankton biomass estimates (i.e., RPDs greater than 50% and Bray-Curtis values greater than 0.5; Appendix 5C, Table 5C-4).

As part of the QA/QC process, the chlorophyll data were reviewed for unusually high or low values. All chlorophyll *a* values were determined to be consistent with previous years.

5.4 Results

Appendix 5C contains detailed results from all sampling events for all components of the plankton program, as follows:

- Appendix 5C, Table 5C-5 – chlorophyll *a* and *c* concentrations;
- Appendix 5C, Tables 5C-6, 5C-7, and 5C-8 – phytoplankton taxonomic data (i.e., abundance and biomass); and,
- Appendix 5C, Tables 5C-9, 5C-10, and 5C-11 – zooplankton taxonomic data (i.e., abundance and biomass).

5.4.1 Light Assessment

The main basin of Snap Lake had the deepest mean (± SD) euphotic zone (greater than 1% surface irradiance) depth measured by both the LI-COR light meter (12 ± 3 m) and Secchi depth (11 ± 1 m) followed by Northeast Lake (11 ± 2 m and 9 ± 1 m, respectively), Lake 13 (9 ± 0m for both methods), and the northwest arm of Snap Lake (9 ± 0 m and 7 ± 0 m, respectively; Table 5-5). The euphotic zones in Snap Lake, Northeast Lake, and Lake 13 extended to the bottom depths, with the exception of the deep stations in the main basin (Station SNAP02-20e) and the northwest arm (Station SNAP20B) of Snap Lake. Overall, Secchi depth estimates of the euphotic zone were consistently lower than those measured with the LI-COR light meter (Table 5-5; Appendix 5A, Table 5A-1).

The vertical light profiles demonstrated that the deepest mean euphotic zone depth occurred in the main basin of Snap Lake at 12 m (Table 5-5). Within the main basin of Snap Lake, the deepest euphotic zones occurred at stations SNAP11A and SNAP02-20e, ranging from 14 to 16 m. Although Station SNAP08 had the shallowest recorded euphotic zone depth, irradiance at Station SNAP08 reached to the bottom of the water column (8 to 9 m). Irradiance curves for plankton stations in the main basin of Snap Lake are characteristic of mesotrophic systems, with the exception of the July irradiance curve at Station SNAP08, which was most similar to that of an oligotrophic lake (Appendix 5A, Figure 5A-3). Typically, a higher percentage of surface irradiance at each depth was observed in July compared to August or September, with the exception of Station SNAP02-20e (diffuser station), where little variation was observed between sampling periods (Appendix 5A, Figure 5A-3). Monthly variation was highest at stations SNAP06 and SNAP08 in the main basin of Snap Lake (Appendix 5A, Figure 5A-3).

The mean euphotic zone in the northwest arm of Snap Lake reached to about 9 ± 0 m. In this area of Snap Lake, irradiance reached to the bottom depths of the water column at all stations, except at Station SNAP20B, which had a maximum depth of 30 m, but a euphotic zone depth of 8 m in July and 9 m in August (Table 5-5; Appendix 5A, Figure 5A-4). Little to no monthly variation between July and August was observed at the plankton stations in the northwest arm. In general, the irradiance curves observed in the northwest arm of Snap Lake were characteristic of mesotrophic systems (Appendix 5A, Figure 5A-4).

The euphotic zones in the two reference lakes, Northeast Lake and Lake 13, reached mean depths of approximately 11 ± 2 m and 9 ± 0 m, respectively, equal to the maximum water depth (Table 5-5). In both reference lakes, monthly variation in irradiance curves was observed (Appendix 5A, Figures 5A-5 to 5A-6); however, this monthly variation was inconsistent among stations. In general, irradiance curves in both reference lakes were characteristic of mesotrophic lakes, with the exception of Station NEL02 in Northeast Lake in September, which had an irradiance curve most similar to an oligotrophic lake (Appendix 5A, Figure 5A-5).

Table 5-5 Euphotic Zone Depth within Snap Lake, Northeast Lake, and Lake 13, 2013

Lake and Area	Station	Mean Maximum Depth (m)[(a)]	LI-COR Light Meter				Mean ± SD Secchi Depth Euphotic Zone Depth (m)[(b)]
			Euphotic Zone Depth (m)			Mean ± SD Euphotic Zone Depth (m)	
			July	August	September		
Snap Lake – Main Basin	SNAP02-20e	25	14	16	16	15 ±1	13 ±4
	SNAP03	13	13	12	12	12 ±1	12 ±2
	SNAP06	12	12	12	9	11 ±2	12 ±1
	SNAP11A	16	16	16	14	15 ±1	11 ±3
	SNAP08	9	8	8	9	8 ±1	9 ±1
	Area Mean	n/a	n/a	n/a	n/a	12 ±3	11 ±1
Snap Lake – Northwest Arm	SNAP29	7	7	7	-	7 ±0	7 ±1
	SNAP23	12	11	11	-	11 ±0	7 ±2
	SNAP02A	9	9	9	-	9 ±0	7 ±1
	SNAP20B	30	9	8	-	9 ±1	7 ±1
	Area Mean	n/a	n/a	n/a	n/a	9 ±0	7 ±0
Northeast Lake	NEL01	12	11	11	12	11 ±1	9 ±1
	NEL02	12	11	12	11	11 ±1	10 ±2
	NEL03	9	9	9	8	9 ±1	9 ±0
	NEL04	12	11	12	9	11 ±1	10 ±2
	NEL05	12	10	12	10	11 ±1	9 ±1
	Area Mean	n/a	n/a	n/a	n/a	11 ±2	9 ±1
Lake 13	LK13-01	12	8	11	12	10 ±2	9 ±2
	LK13-02	10	6	10	7	8 ±2	8 ±2
	LK13-03	11	8	11	8	9 ±2	9 ±2
	LK13-04	11	8	10	10	9 ±1	9 ±2
	LK13-05	15	7	12	11	10 ±3	9 ±2
	Area Mean	n/a	n/a	n/a	n/a	9 ±0	9 ±0

a) Mean maximum water depth was based on the maximum water depth measured among all three sampling periods.

b) Mean Secchi depth was based on the mean Secchi depth measurements among all three sampling periods.

m = metre; SD = standard deviation; n/a = not applicable; - = not measured due to LI-COR meter malfunction; ± = plus or minus; SNAP = Snap lake; LK13 = Lake 13; NEL = Northeast Lake.

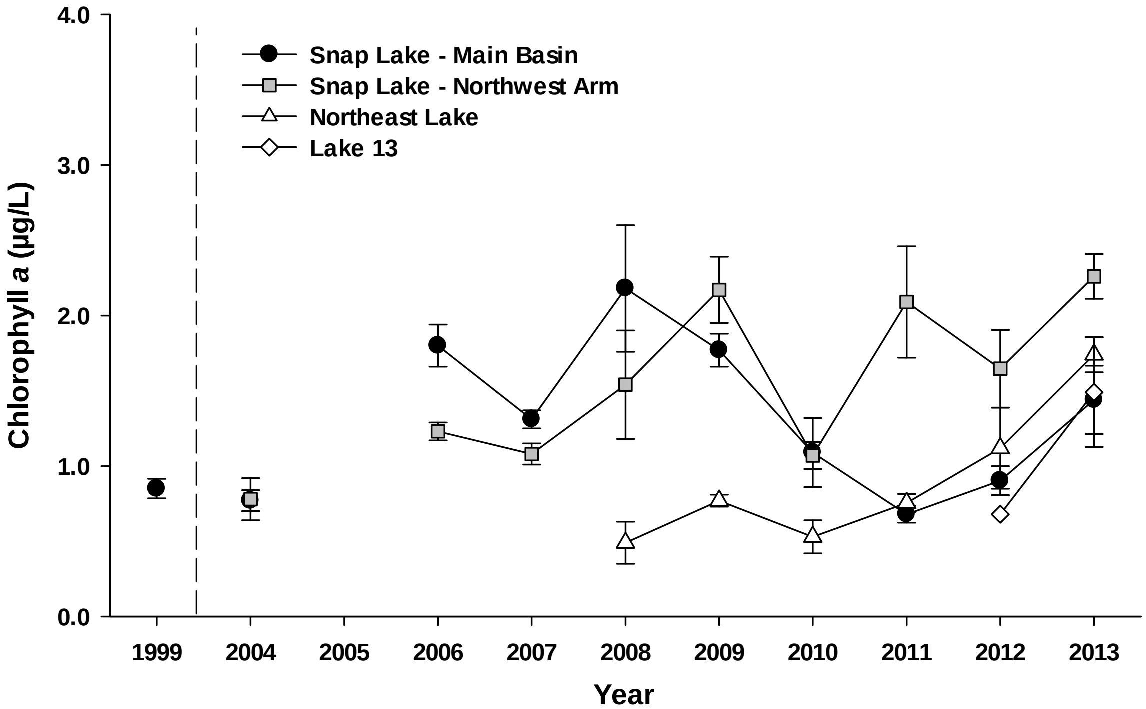
5.4.2 Chlorophyll *a* and *c* Concentrations

Mean (± SE) annual chlorophyll *a* concentrations have varied over time in Snap Lake (Figure 5-5). An increase from baseline concentrations (i.e., 2004) was observed in the main basin of Snap Lake between 2004 and 2008. Following 2008, chlorophyll *a* concentrations steadily declined in the main basin of Snap Lake, returning to near baseline concentrations in 2011. Since that time, an increasing trend in chlorophyll *a* concentrations has been observed; in 2013 chlorophyll *a* concentrations were approximately double the baseline concentration. In general, chlorophyll *a* concentrations have increased from baseline in the northwest arm of Snap Lake, with the exception of 2010, when concentrations decreased by about 50%. Since 2009, chlorophyll *a* concentrations have been higher in the northwest arm compared to the main basin of Snap Lake. Chlorophyll *a* concentrations have also increased over time in Northeast Lake (2008 to 2013) and in Lake 13 (2012 and 2013).

In 2013, chlorophyll *a* concentrations in the main basin of Snap Lake ranged from 0.24 to 3.11 µg/L (Appendix 5C, Table 5C-5). Chlorophyll *a* concentrations generally peaked in September in the main basin of Snap Lake, although chlorophyll *a* concentrations at Station SNAP08 were comparable in July and September (Figure 5-6). Chlorophyll *a* concentrations were highest in the northwest arm of Snap Lake, ranging from 1.16 to 4.37 µg/L and were more variable with no distinct seasonal pattern, Chlorophyll *a* concentrations ranged from 0.55 to 3.26 µg/L in Northeast Lake. Seasonal peaks in chlorophyll *a* concentrations occurred in July at all stations in Northeast Lake, with substantially lower chlorophyll *a* concentrations occurring in August. In Lake 13, chlorophyll *a* concentrations ranged from 0.59 to 3.38 µg/L and were elevated in July and September, with three of the five stations having clear peak concentrations in July. Although chlorophyll concentrations were highest at the diffuser station (SNAP02-20e), overall there were no clear spatial patterns chlorophyll *a* concentrations in 2013 in relation to proximity to the diffuser in Snap Lake (Figure 5-6; Appendix 5C, Table 5C-5).

Chlorophyll *c* concentrations ranged from less than 0.004 to 0.25 µg/L in the main basin of Snap Lake compared with the northwest arm of Snap Lake, where chlorophyll *c* concentrations generally ranged from less than 0.004 to 0.44 µg/L. Relatively high chlorophyll *c* concentrations occurred at Station SNAP29 in July (5.12 µg/L and 4.64 µg/L); there was no apparent reason for these high concentrations as phytoplankton biomass was not elevated at this station (see Section 5.4.3.4). Chlorophyll *c* concentrations ranged from less than 0.004 to 0.19 µg/L in Northeast Lake and from less than 0.004 to 0.07 µg/L in Lake 13. No distinct seasonal patterns were observed in chlorophyll *c* concentrations in Snap Lake or Northeast Lake; however, in Lake 13 chlorophyll *c* concentrations consistently increased in September, with the exception of LK13-01, which had high chlorophyll *c* concentrations in both July and September (Figure 5-7). There were no clear spatial patterns of chlorophyll *c* concentrations in 2013 in Snap Lake in relation to proximity to the diffuser (Figure 5-7).

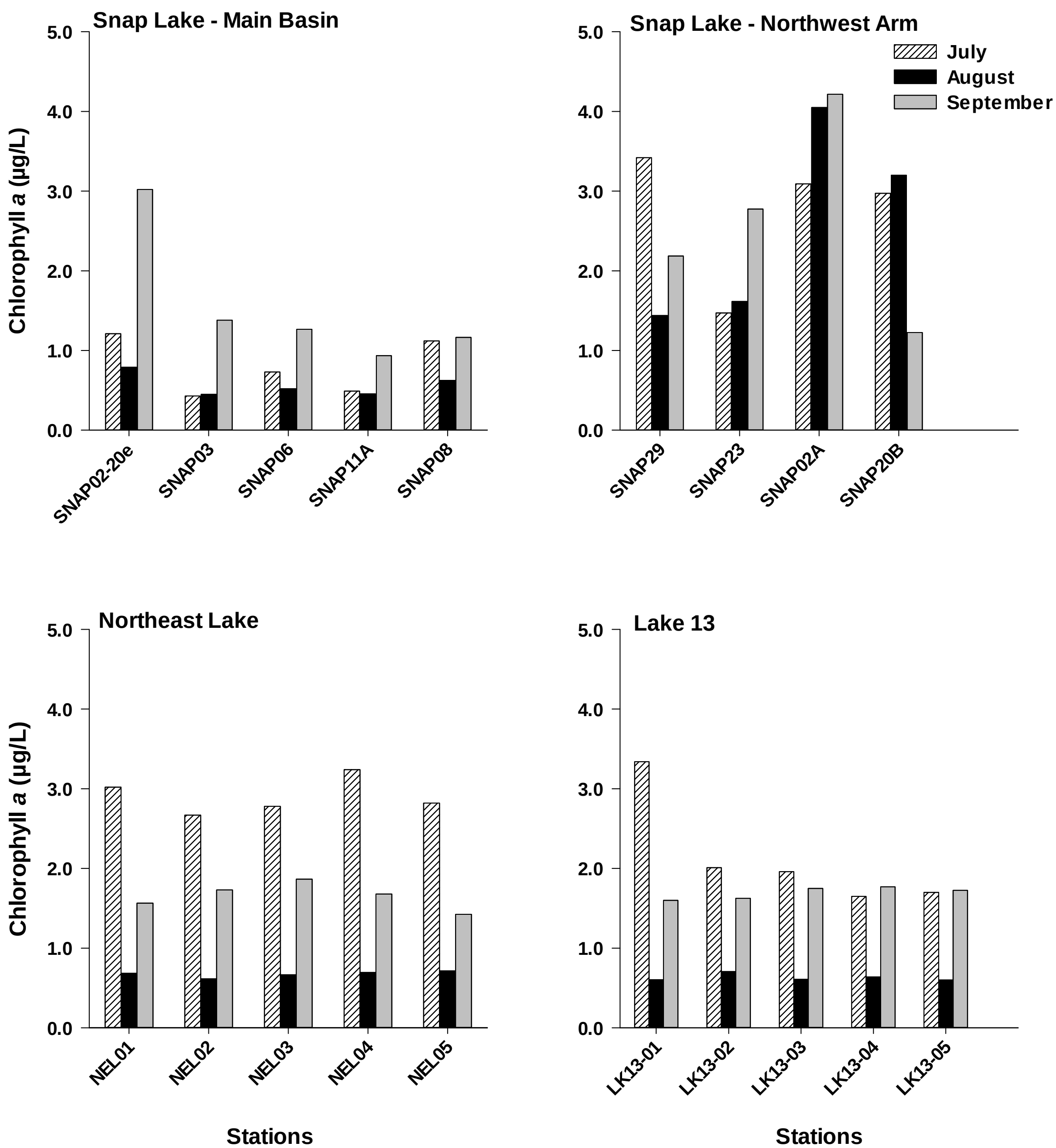
Figure 5-5 Temporal Trends in Mean Annual Chlorophyll *a* in the Main Basin and Northwest Arm of Snap Lake, Northeast Lake, and Lake 13



Notes: Error bars represent standard error of the seasonal means. The 1999 baseline data were not separated into the northwest arm and main basin of Snap Lake (De Beers 2002). Chlorophyll *a* sampling did not occur in Northeast Lake until July 2008 and did not include an August sampling event until 2011. Chlorophyll *a* sampling did not occur in Lake 13 until August 2012 and did not include July and September sampling events until 2013. Chlorophyll *a* data from 2005 were determined to be outliers; therefore, these data were omitted (De Beers 2011). Chlorophyll *a* was not collected at SNAP08 in July 2012 and data from SNAP06 in August 2012 were determined to be outliers and were removed from the plot (De Beers 2013). The vertical dashed bar represents a break in the time series and change in sampling methods.

µg/L = micrograms per litre.

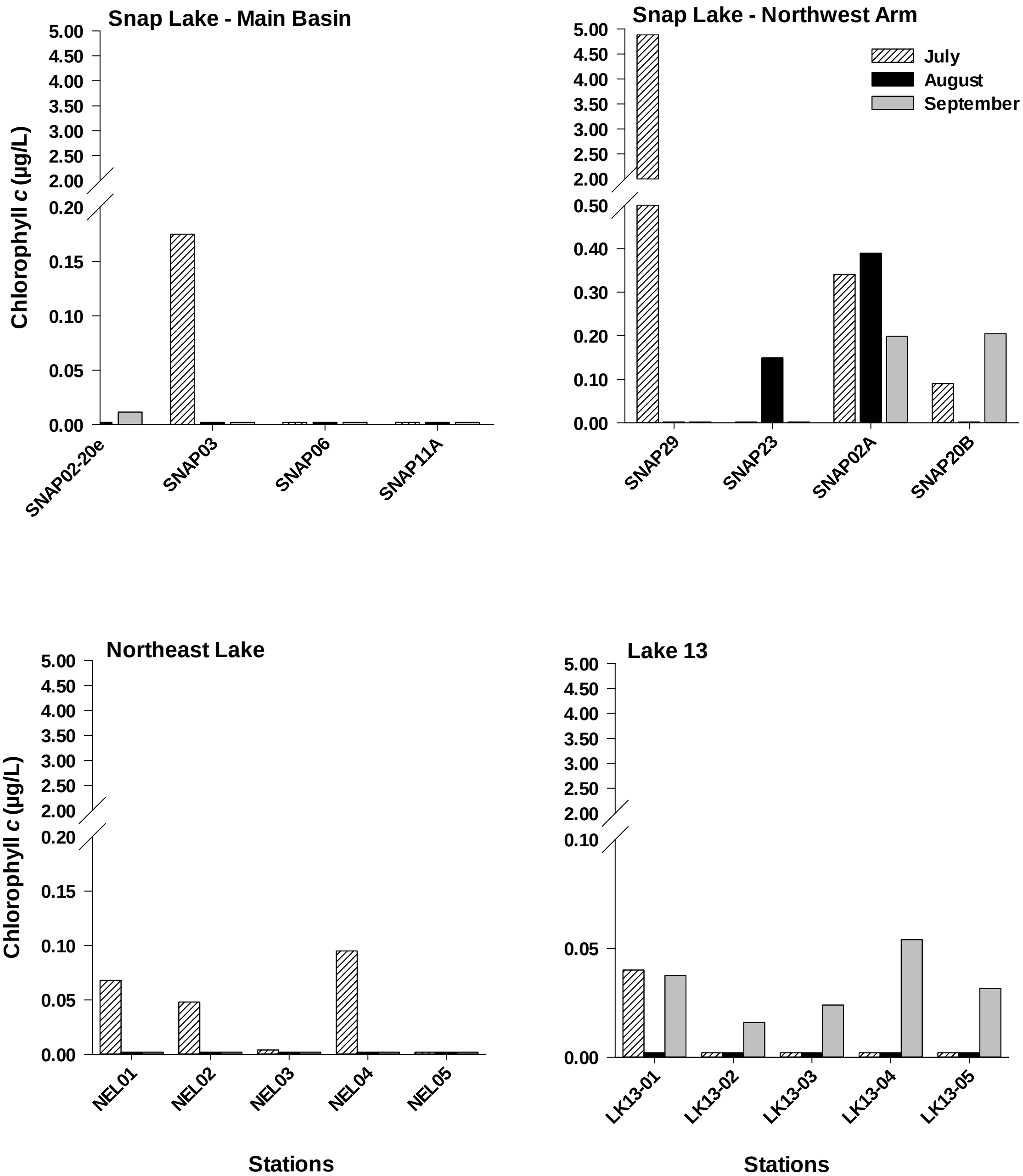
Figure 5-6 Spatial and Seasonal Trends in Chlorophyll *a* Concentrations in the Main Basin and Northwest Arm of Snap Lake, Northeast Lake, and Lake 13, 2013



Notes: Stations are arranged from closest to the diffuser (SNAP02-20e) to farthest from the diffuser (SNAP08) in the main basin and from closest to the narrows to the main basin (SNAP29) to farthest from the narrows (SNAP20B) in the northwest arm of Snap Lake.

µg/L = micrograms per litre.

Figure 5-7 Spatial and Seasonal Trends in Chlorophyll *c* Concentrations in the Main Basin and Northwest Arm of Snap Lake, Northeast Lake, and Lake 13, 2013



Notes: Stations are arranged from closest to the diffuser (SNAP02-20e) to farthest from the diffuser (SNAP08) in the main basin and from closest to the narrows to the main basin (SNAP29) to farthest from the narrows (SNAP20B) in the northwest arm of Snap Lake.

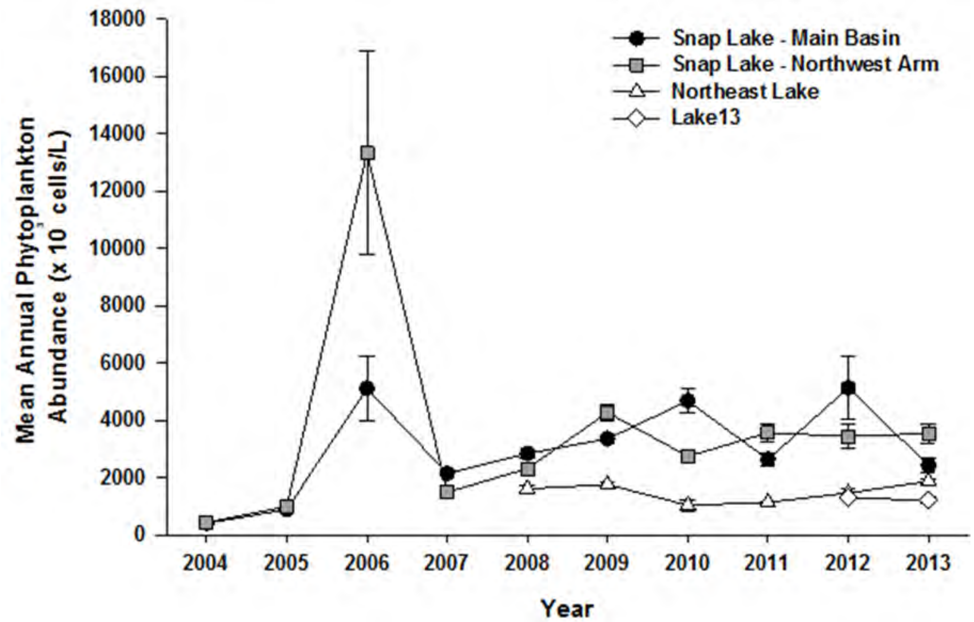
µg/L = micrograms per litre.

5.4.3 Phytoplankton Community

5.4.3.1 Total Phytoplankton Abundance

Mean annual total phytoplankton abundance has increased since baseline (i.e., 2004) in the main basin and northwest arm of Snap Lake (Figure 5-8). Mean (± SE) annual total phytoplankton abundance at baseline was 425,000 ± 17,000 cells per litre (cells/L) and, with the exception of 2006, gradually increased in both areas of Snap Lake until 2009. In 2006, total phytoplankton abundance showed a sharp increase in both the main basin (5,127,000 ± 1,142,000 cells/L) and the northwest arm of Snap Lake (13,345,000 ± 3,531,000 cells/L). Since 2009, total phytoplankton abundance in both areas of Snap Lake has remained relatively constant, fluctuating around 3,000,000 cells/L. In 2013, mean annual total phytoplankton abundance was higher in the northwest arm of Snap Lake (3,556,000 ± 352,000 cells/L) than in the main basin (2,443,000 ± 953,000 cells/L). Mean annual total phytoplankton abundance has remained consistent between 2008 and 2013 in the Northeast Lake and has remained below values observed in Snap Lake. Mean annual phytoplankton abundance in Lake 13 was lower than values observed in Snap Lake and Northeast Lake in 2012 and 2013.

Figure 5-8 Temporal Trends in Mean Annual Phytoplankton Abundance in the Main Basin and Northwest Arm of Snap Lake, Northeast Lake, and Lake 13.



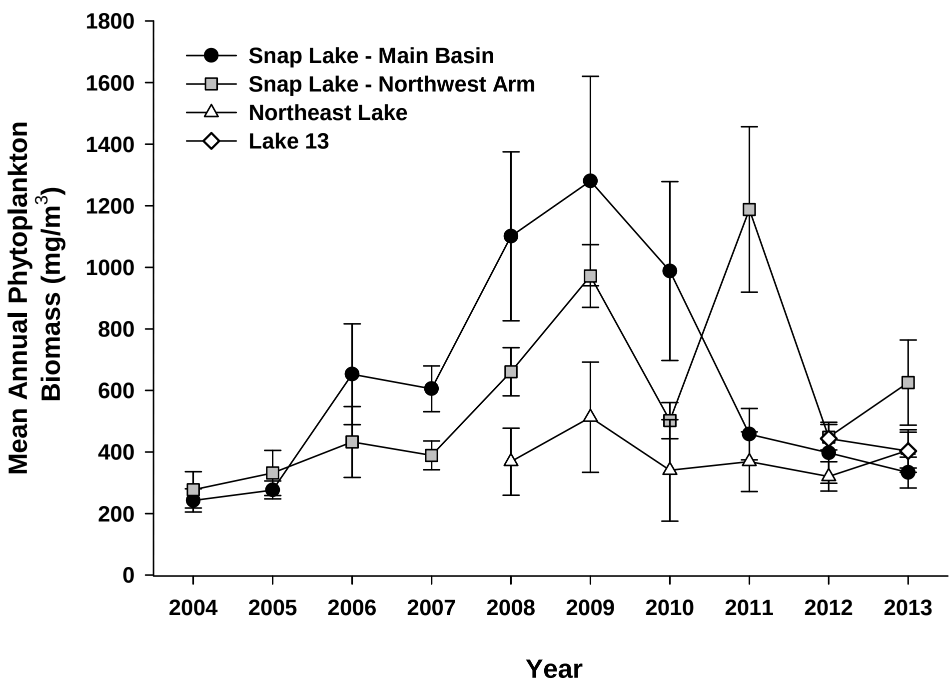
Notes: Error bars represent standard error of the seasonal means. Sampling did not occur in Northeast Lake until July 2008 and did not include an August sampling event until 2011. Sampling did not occur in Lake 13 until 2012 and did not include July and September sampling events until 2013.

cells/L = cells per litre.

5.4.3.2 Phytoplankton Biomass

Mean annual total phytoplankton biomass has varied over time in the main basin and northwest arm of Snap Lake (Figure 5-9). In the main basin of Snap Lake, total phytoplankton biomass increased from 2004 to 2009 and decreased from 2009 to 2013, returning to near the baseline (i.e., 2004) value. In 2013, mean (± SE) annual total phytoplankton biomass in the main basin was 333 ± 30 milligrams per cubic metre (mg/m^3), which is approximately 1.4 times higher than the baseline value in 2004 (243 ± 38 mg/m^3). Mean annual total phytoplankton biomass in the northwest arm of Snap Lake was consistently lower than in the main basin prior to 2011. In 2011, total phytoplankton biomass increased in the northwest arm and decreased in the main basin; since that time, total phytoplankton biomass has been higher in the northwest arm compared to the main basin of Snap Lake. There has been little change in the mean annual total phytoplankton biomass in Northeast Lake since sampling began in 2008, with total phytoplankton biomass values remaining close to the Snap Lake baseline value. Mean annual total phytoplankton biomass in Lake 13 in 2013 (403 ± 69 mg/m^3) was similar to total phytoplankton biomass observed in Northeast Lake (406 ± 58 mg/m^3). Mean annual total phytoplankton biomass was lower in the main basin (333 ± 38 mg/m^3) and higher in the northwest arm (626 ± 138 mg/m^3) of Snap Lake compared to Lake 13.

Figure 5-9 Temporal Trends in Mean Annual Phytoplankton Biomass in the Main Basin and Northwest Arm of Snap Lake, Northeast Lake, and Lake 13.



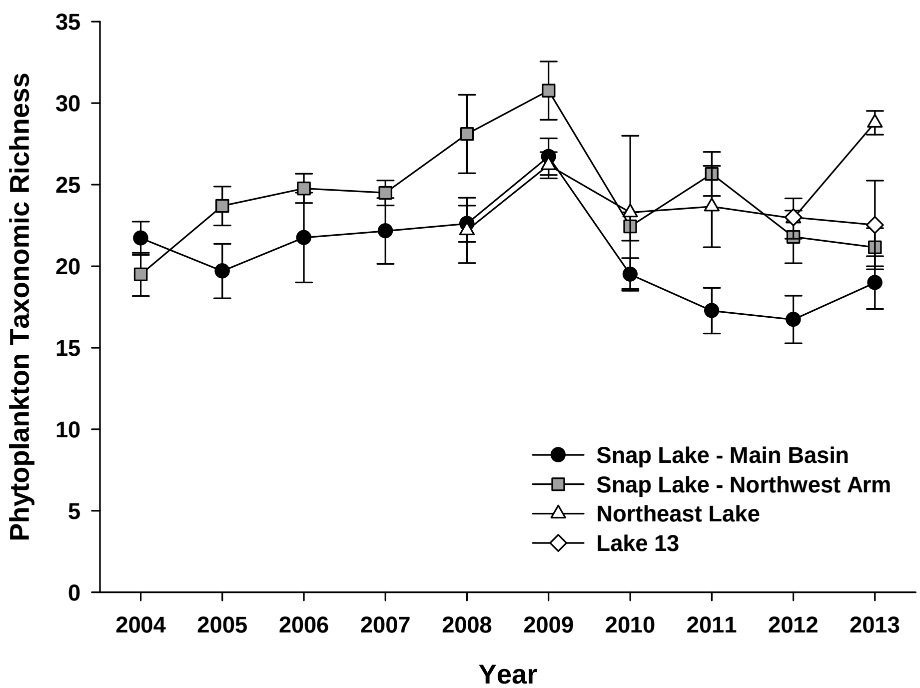
Notes: Error bars represent standard error of the seasonal mean. Sampling did not occur in Northeast Lake until July 2008 and did not include an August sampling event until 2011. Sampling did not occur in Lake 13 until 2012 and did not include July and September sampling events until 2013. The 1999 baseline data were not separated into the northwest arm and main basin of Snap Lake (De Beers 2002). The vertical dashed bar represents a break in the time series and change in sampling methods.

mg/m^3 = milligrams per cubic metre.

5.4.3.3 Phytoplankton Taxonomic Richness

Mean annual phytoplankton taxonomic richness, has varied over time in the three study lakes (Figure 5-10). In general, there was a slight increase in phytoplankton taxonomic richness in Snap Lake between 2004 and 2009. Since 2009, phytoplankton taxonomic richness has declined in both areas of Snap Lake. Phytoplankton taxonomic richness has been consistently higher in the northwest arm compared to the main basin although, in 2013, mean (± SE) annual phytoplankton taxonomic richness values were similar between the main basin (19 ± 2) and the northwest arm (20 ± 1) of Snap Lake. The 2013 values were comparable to the 2004 baseline values in the main basin (22 ± 1) and northwest arm of Snap Lake. Little change had been observed in phytoplankton taxonomic richness in Northeast Lake until 2013, when there was an increase. In 2013, phytoplankton taxonomic richness values were higher in Northeast Lake and Lake 13 compared to the main basin and northwest arm of Snap Lake.

Figure 5-10 Temporal Trends in Mean Annual Phytoplankton Taxonomic Richness in the Main Basin and Northwest Arm of Snap Lake, Northeast Lake, and Lake 13.



Notes: Phytoplankton taxonomic richness is based on a genus-level assessment. Error bars represent standard error of the seasonal mean. Sampling did not occur in Northeast Lake until July 2008 and did not include an August sampling event until 2011. Sampling did not occur in Lake 13 until 2012 and did not include July and September sampling events until 2013.

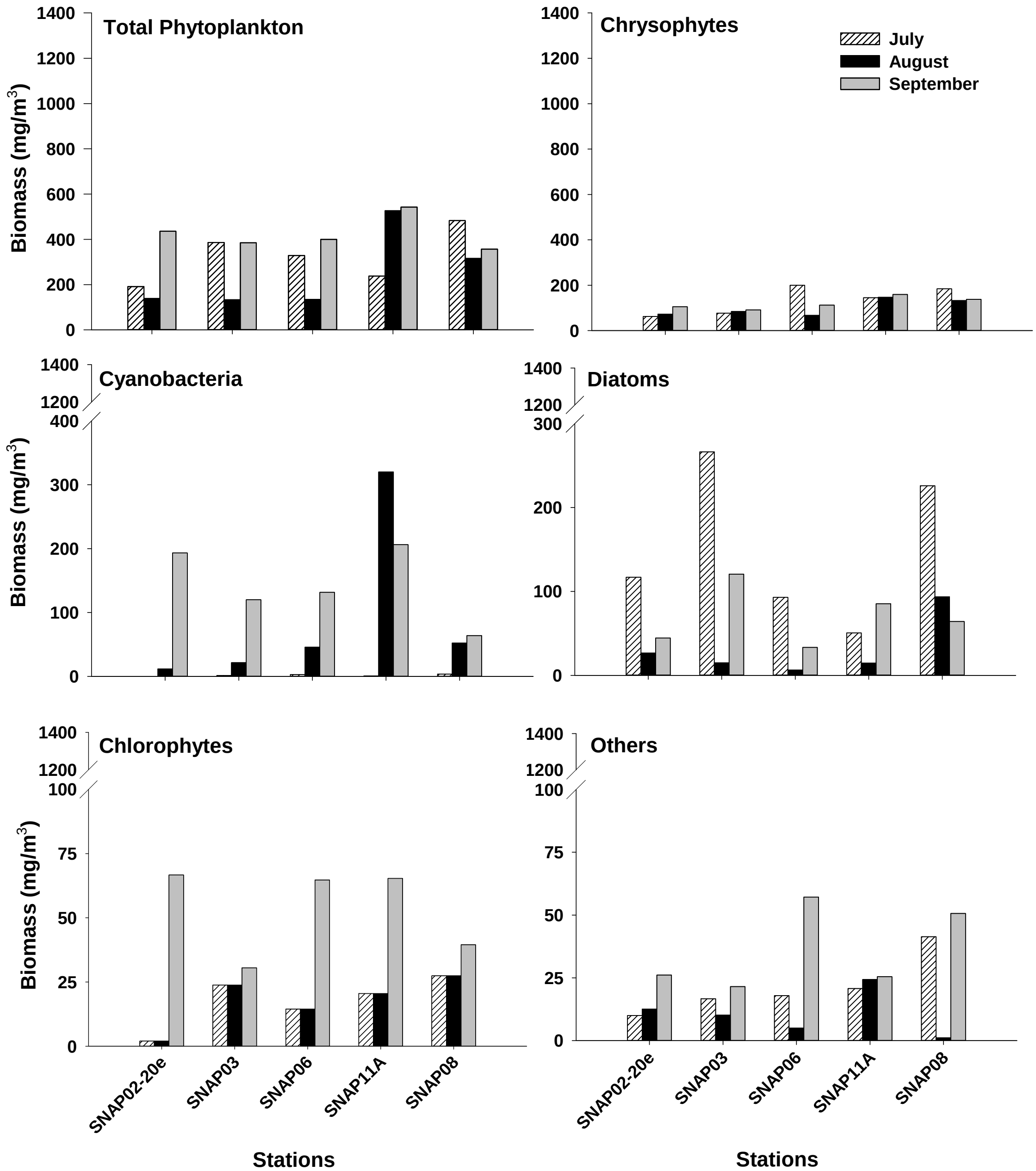
5.4.3.4 2013 Seasonal and Spatial Trends

Total phytoplankton biomass generally peaked in September at all stations in the main basin and northwest arm of Snap Lake and Northeast Lake (Figures 5-11, 5-12, and 5-13), with two exceptions: Station SNAP08; and, Station SNAP20B. Total phytoplankton biomass peaked in July at SNAP08 and in August at SNAP20B. Seasonal peaks in phytoplankton biomass in Lake 13 varied from station to station, with no consistent seasonal peaks (Figure 5-14).

Seasonal peaks in biomass of the major phytoplankton groups varied from group-to-group and from lake-to-lake (Figures 5-11, 5-12, 5-13, and 5-14). In the main basin of Snap Lake and Northeast Lake, there were no consistent seasonal peaks in chrysophyte biomass; however, at the majority of stations in the northwest arm of Snap Lake and in Lake 13, seasonal peaks occurred in August. Seasonal peaks in cyanobacteria biomass generally occurred in September in Snap Lake, Northeast Lake, and Lake 13, with the exception of SNAP11A and LK13-03 where cyanobacterial biomass peaks occurred in August. Seasonal peaks in diatom biomass generally occurred in all three lakes in July or September, while seasonal peaks in chlorophyte biomass were observed in August or September. There were no distinct seasonal trends in the 'other' major phytoplankton group.

No consistent spatial patterns in total phytoplankton biomass, in relation to proximity to the diffuser, were observed in Snap Lake in 2013; any spatial variation was likely related to the natural variability in phytoplankton community structure (Figures 5-11 and 5-12). Chrysophyte biomass decreased with distance from the main basin into the northwest arm of Snap Lake (i.e., from SNAP02-20e, SNAP03, SNAP29, and SNAP23; Figures 5-11 and 5-12). No clear spatial patterns were observed in the other major phytoplankton groups in 2013. Spatial variation among stations was observed in Northeast Lake and Lake 13, consistent with the high natural spatial variability observed in phytoplankton community data (Figures 5-13 and 5-14).

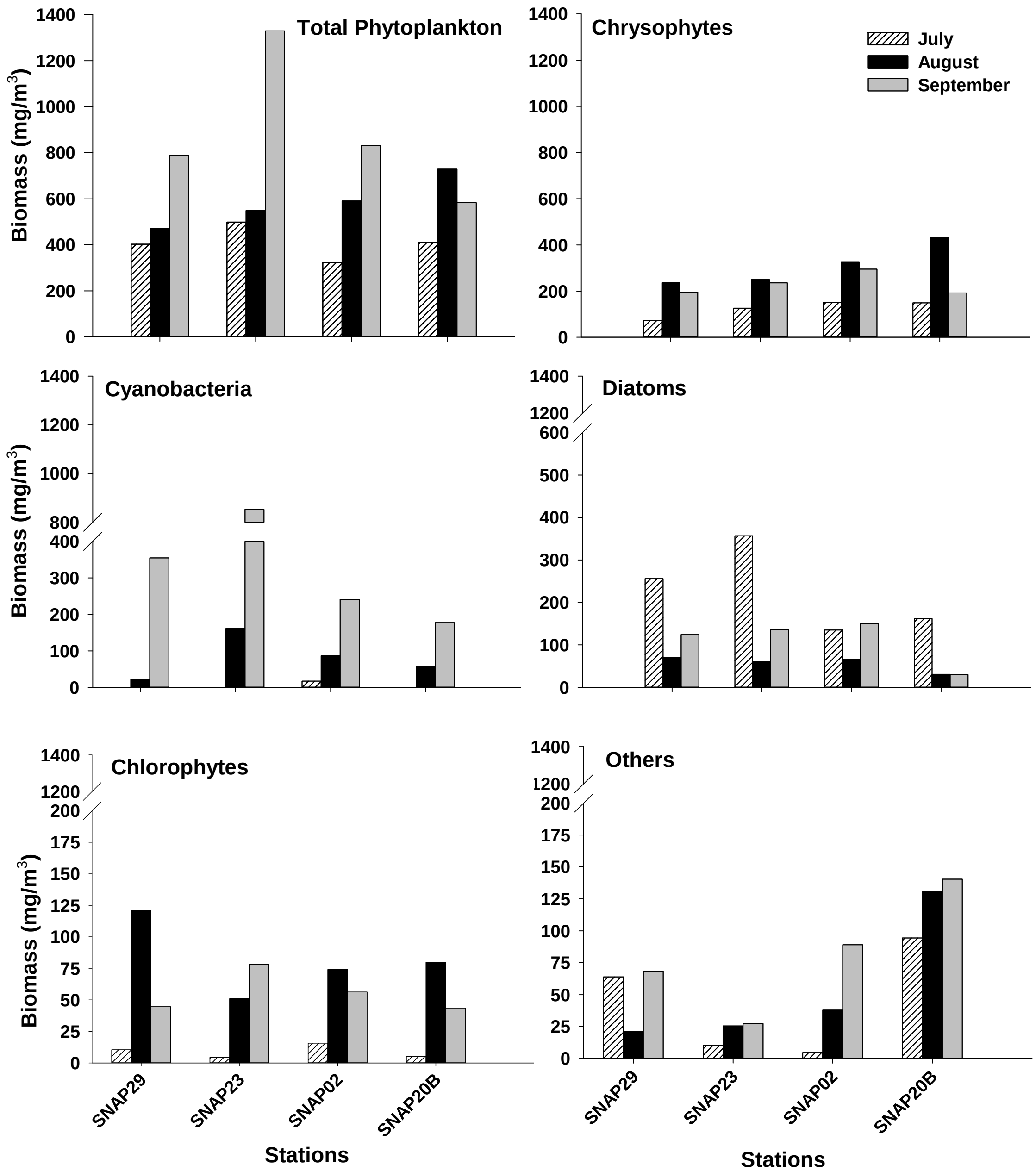
Figure 5-11 Spatial and Seasonal Trends in Biomass of the Major Phytoplankton Groups in the Main Basin of Snap Lake, 2013



Notes: Stations are arranged from closest to the diffuser (SNAP02-20e) to farthest from the diffuser (SNAP08). The “Others” taxonomic group includes euglenoids, dinoflagellates, and xanthophytes.

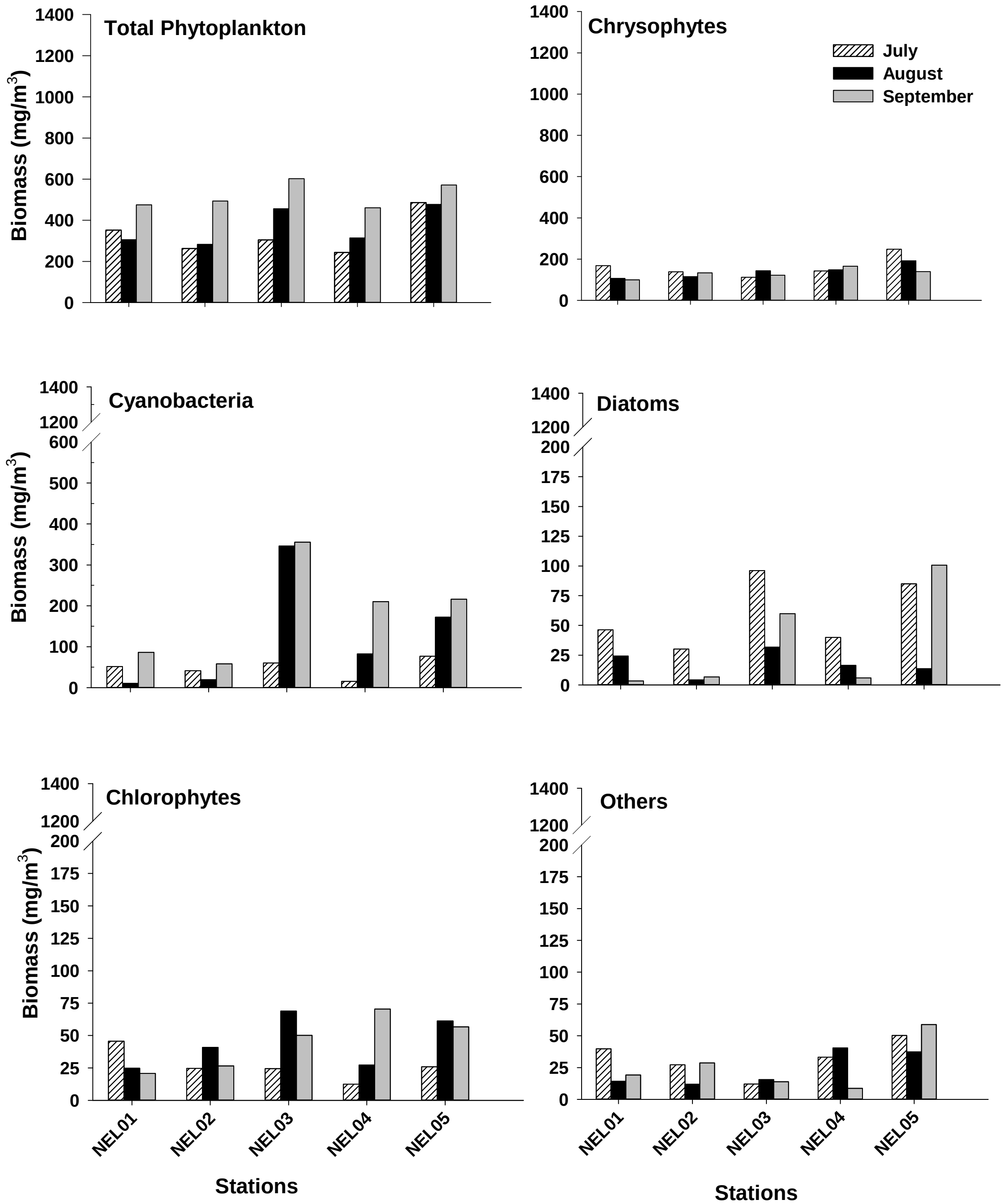
mg/m^3 = milligrams per cubic metre.

Figure 5-12 Spatial and Seasonal Trends in Biomass of the Major Phytoplankton Groups in the Northwest Arm of Snap Lake, 2013



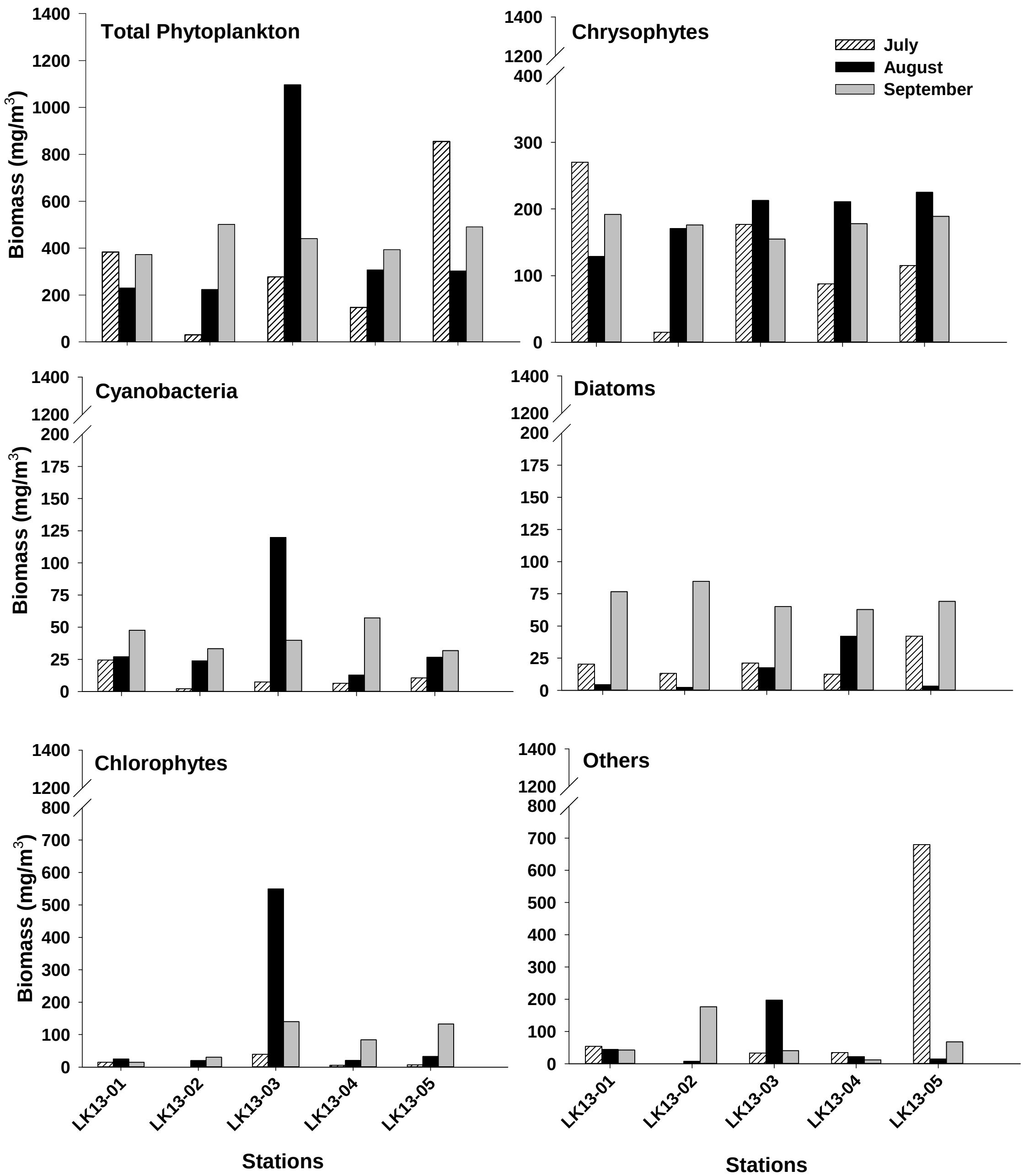
Notes: Stations are arranged from closest to the narrows to the main basin (SNAP29) to farthest from the narrows (SNAP20B) in the northwest arm of Snap Lake. The “Others” taxonomic group includes euglenoids, dinoflagellates, and xanthophytes.
mg/m^3 = milligrams per cubic metre.

Figure 5-13 Spatial and Seasonal Trends in Biomass of the Major Phytoplankton Groups in Northeast Lake, 2013



Note: The "Others" taxonomic group includes euglenoids, dinoflagellates, and xanthophytes .
mg/m^3 = milligrams per cubic metre.

Figure 5-14 Spatial and Seasonal Trends in Biomass of the Major Phytoplankton Groups in Lake 13, 2013



Note: The "Others" taxonomic group includes euglenoids, dinoflagellates, and xanthophytes
mg/m^3 = milligrams per cubic metre.

5.4.3.5 Phytoplankton Community Composition

In terms of relative abundance, chrysophytes dominated the phytoplankton community in the main basin of Snap Lake from 2004 to 2009, with the exception of 2006, when chlorophytes became the dominant group (Figure 5-15). From 2009 to 2012, diatom abundance increased and the phytoplankton community in the main basin of Snap Lake shifted to a diatom-chrysophyte co-dominated community. In 2013, a decrease in diatom dominance and an increase in chrysophyte dominance occurred, with chrysophytes representing 60% of the phytoplankton community in the main basin of Snap Lake. Chrysophytes have consistently dominated relative abundance in the northwest arm of Snap Lake from 2004 to 2013, with the exception of 2006 when chlorophytes became the dominant group. Relative phytoplankton abundance in Northeast Lake has differed from that in Snap Lake. In Northeast Lake, cyanobacteria were the dominant group from 2008 to 2011, with chrysophytes as the sub-dominantgroup. After 2011, chrysophyte abundance increased; by 2013, chrysophytes were the dominant group in Northeast Lake (48%). Relative phytoplankton abundance in Lake 13 has also been chrysophyte-dominated, accounting for 65% of the phytoplankton community in 2013.

In terms of relative biomass, chrysophytes and cyanobacteria co-dominated the phytoplankton community in the main basin of Snap Lake in 2004 and 2005 (Figure 5-16). Cyanobacteria were the dominant group in 2006. From 2007 to 2012, the relative proportion of cyanobacteria biomass decreased and the community shifted to a diatom-chrysophyte co-dominated community; however, the relative proportion of cyanobacteria increased from 2.5% in 2012 to 20% in 2013. The increase in cyanobacteria biomass in 2013 was caused by an increase in a colonial taxa: *Aphanocapsa delicatissima* (Appendix 5C, Table 5C-6).

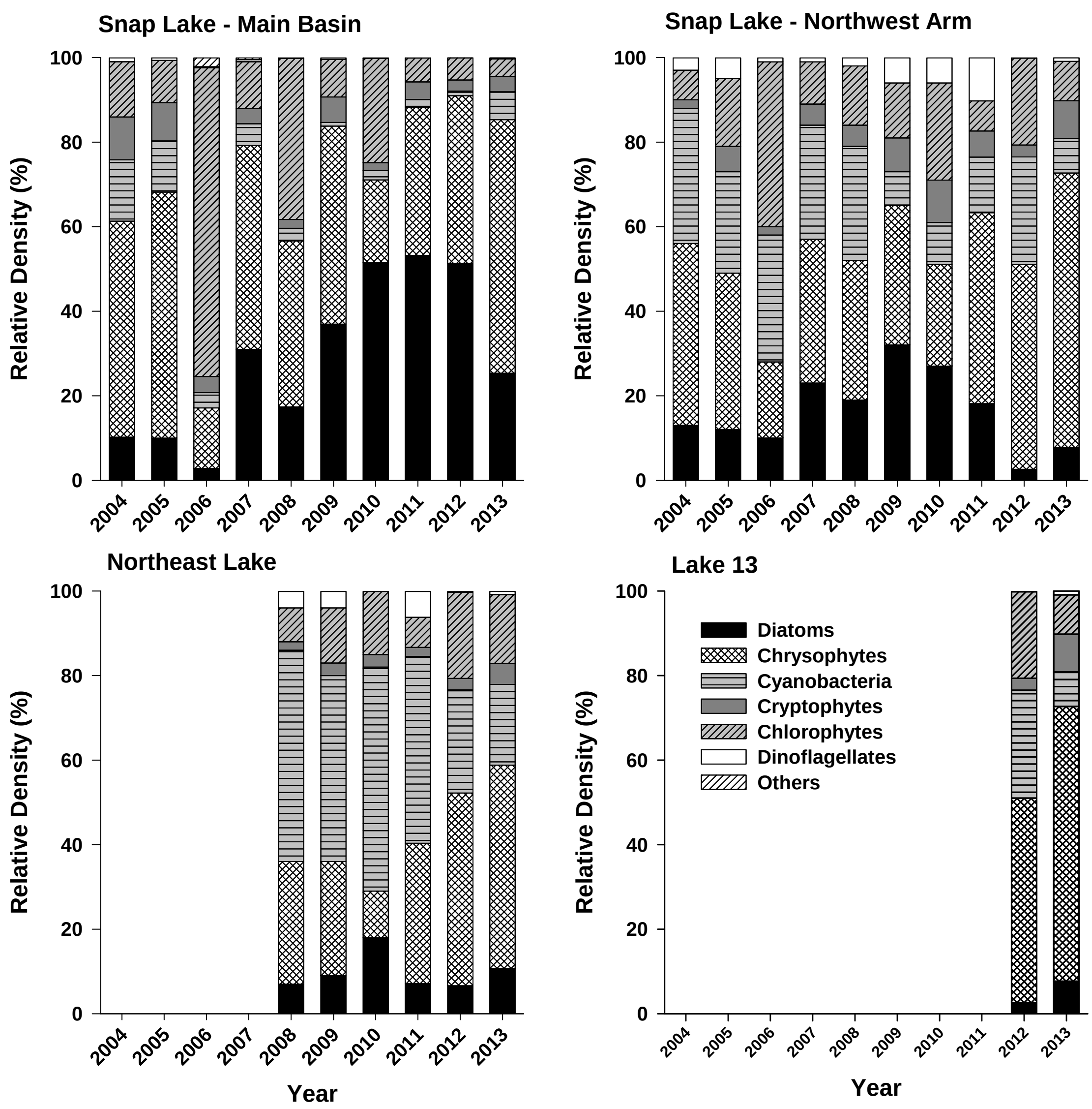
The contribution of chrysophyte biomass to the phytoplankton assemblage in the main basin of Snap Lake varied from 2008 to 2013, shifting from dominance (2004 and 2005) to co-dominance (2007 and 2008), to sub-dominance (from 2009 to 2012) back to co-dominance in 2013. Diatom biomass increased from baseline conditions in 2004 and accounted for approximately 60% of total phytoplankton biomass in 2009 and 2011. Relative diatom biomass decreased in 2012 and 2013, accounting for about 25% of total phytoplankton biomass in 2013.

Phytoplankton relative biomass has varied over time in the northwest arm of Snap Lake (Figure 5-16). The community has been mainly chrysophyte-dominated from 2004 to 2013, with the exception of 2006 when the phytoplankton community was chlorophyte-dominated. Sub-dominance has shifted between diatoms and cyanobacteria from 2004 and 2013. A similar community composition was observed in the northwest arm of Snap Lake as in the main basin in 2013, with chrysophytes forming the dominant group (37%), followed by diatoms (26%), and cyanobacteria (19%).

Phytoplankton relative biomass in Northeast Lake has been consistently cyanobacteria-chrysophyte co-dominated since sampling began in 2008 (Figure 5-16). In 2013, cyanobacteria and chrysophytes continued to co-dominate the community, accounting for 36% and 38% of the relative biomass, respectively. Phytoplankton relative biomass in Lake 13 differed from both Northeast Lake and

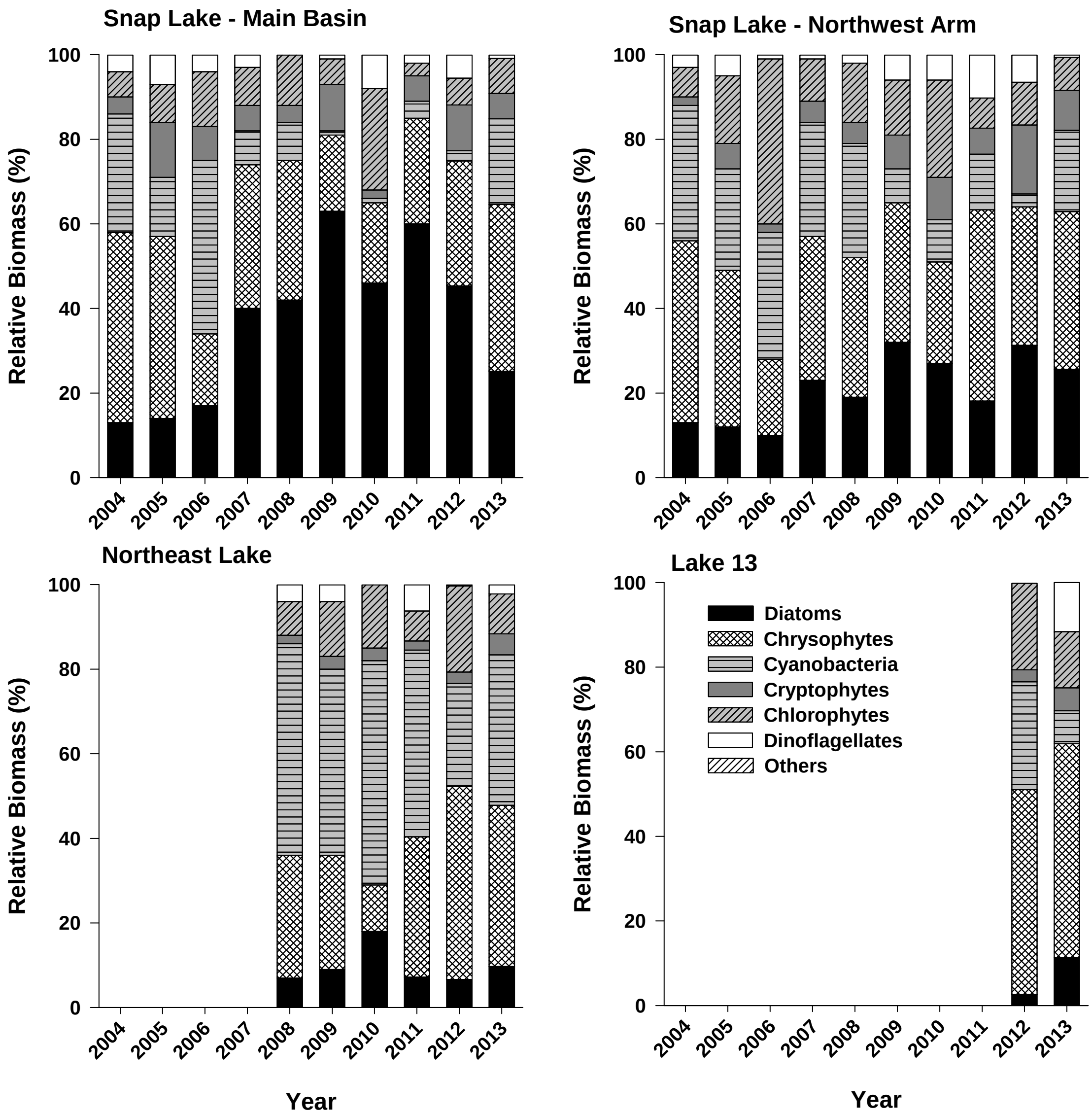
Snap Lake. The phytoplankton community in Lake 13 was chrysophyte-dominated (51%) with sub-dominance by chlorophytes (13%), diatoms (11%), and dinoflagellates (11%). Dinoflagellates, which are present in low numbers (i.e., ≤ 2%) in Snap Lake and Northeast Lake, were not observed in the August 2012 samples collected from Lake 13, but represented 11% of the phytoplankton community in 2013. This was largely related to high dinoflagellate biomass observed at LK13-05 in July 2013 (Figure 5-14).

Figure 5-15 Relative Abundance of Phytoplankton in the Main Basin and Northwest Arm of Snap Lake, Northeast Lake, and Lake 13



Notes: Sampling did not occur in Northeast Lake until the July 2008 and did not include an August sampling event until 2011. Sampling did not occur in Lake 13 until August 2012 and did not include July and September sampling events until 2013. The “Others” taxonomic group includes euglenoids, dinoflagellates, and xanthophytes.
% = percent.

Figure 5-16 Relative Biomass of Phytoplankton in the Main Basin and Northwest Arm of Snap Lake, Northeast Lake, and Lake 13.



Notes: Sampling did not occur in Northeast Lake until the July 2008 and did not include an August sampling event until 2011. Sampling did not occur in Lake 13 until August 2012 and did not include July and September sampling events until 2013.

“Others” were present in small numbers but do not comprise enough biomass to be visible on the plots. The “Others” taxonomic group includes euglenoids, dinoflagellates, and xanthophytes.

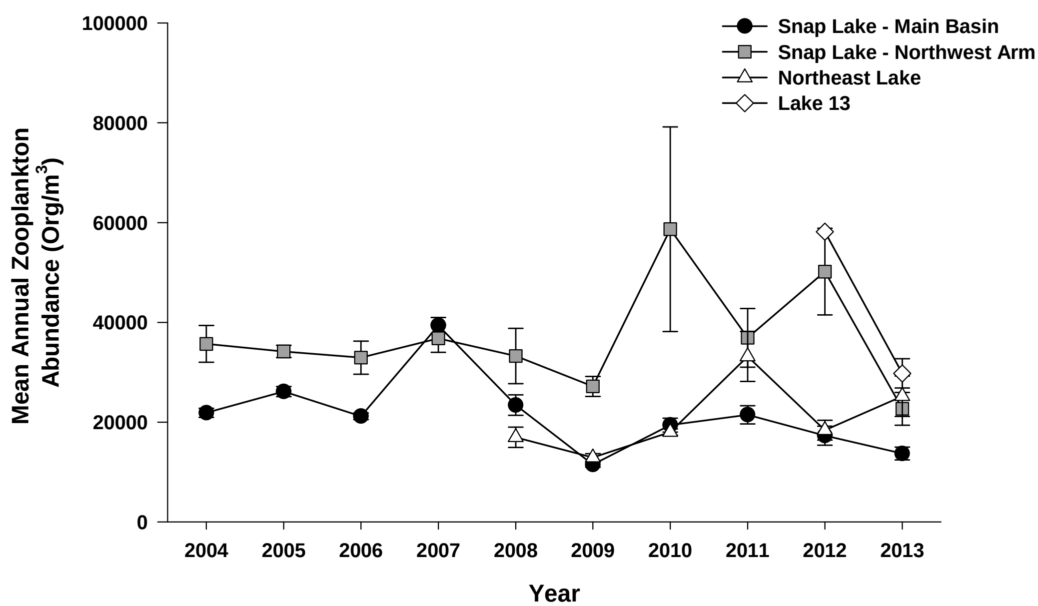
% = percent.

5.4.4 Zooplankton Community

5.4.4.1 Zooplankton Abundance

Mean (± SE) annual total zooplankton abundance varied in Snap Lake over time (Figure 5-17). In the main basin of Snap Lake, mean annual total zooplankton abundance was consistent between 2004 and 2006, followed by a peak in abundance in 2007. After a decrease between 2007 and 2009, mean annual total zooplankton abundance has remained relatively consistent (11,450 to 21,462 organisms per cubic metre [org/m^3]), although slightly below the mean annual baseline value of 21,871 ± 924 org/m^3 in 2004. Mean annual total zooplankton abundance in the northwest arm of Snap Lake has been consistently higher than in the main basin. Little variation in mean annual total zooplankton abundance was observed between 2004 and 2009 in the northwest arm of Snap Lake. In 2010 and 2012, peaks in mean annual zooplankton abundance were observed in the northwest arm of Snap Lake. In 2013, mean annual total zooplankton abundance in the northwest arm of Snap Lake was 22,672 ± 3,291 org/m^3, which was comparable to baseline values in the main basin. In Northeast Lake, mean annual total zooplankton abundance has exhibited a slight increase over time and, since 2010, has been higher compared to the main basin of Snap Lake. In 2012, the highest mean annual total zooplankton abundance was observed in Lake 13; however, in 2013 there was a decline in mean annual total zooplankton abundance to values similar to Northeast Lake. It is possible that this was a reflection of samples being collected only in August 2012.

Figure 5-17 Temporal Trends in Mean Annual Zooplankton Abundance in the Main Basin and Northwest Arm of Snap Lake, Northeast Lake, and Lake 13



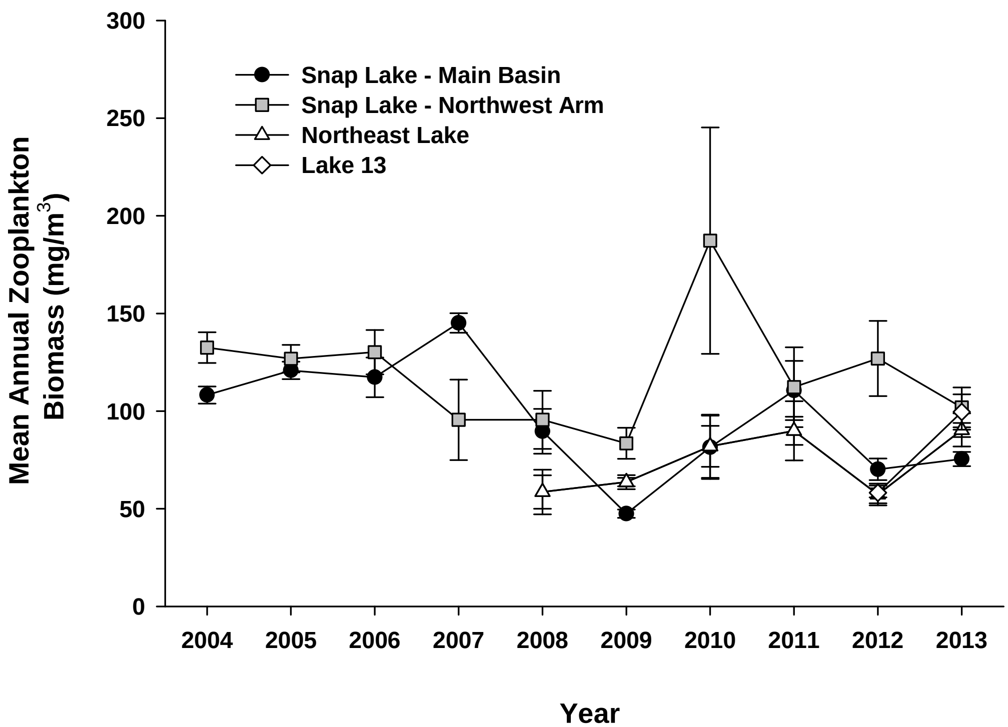
Notes: Error bars represent standard error of the seasonal mean. Sampling did not occur in Northeast Lake until July 2008 and did not include an August sampling until 2011. Sampling did not occur in Lake 13 until August 2012 and did not include July and September sampling until 2013.

org/m^3 = organisms per cubic metre.

5.4.4.2 Zooplankton Biomass

Although mean (± SE) annual total zooplankton biomass has fluctuated over time, overall there has been a decreasing trend in Snap Lake (Figure 5-18). Mean annual total zooplankton biomass in the main basin of Snap Lake has varied from 2004 to 2013. From 2004 to 2007 mean annual total zooplankton biomass increased, and then decreased from 2007 to 2009; from 2008 to 2010 zooplankton biomass remained below baseline, in 2011 (110 ± 15 mg/m^3) it increased to just above the baseline value (108 ± 4 mg/m^3) but decreased again in 2012 and 2013 to below baseline (70 ± 6 mg/m^3 and 75 ± 4 mg/m^3, respectively). In the northwest arm of Snap Lake, mean annual total zooplankton biomass remained stable between 2004 and 2006 and then decreased until 2009. In 2010, mean annual zooplankton biomass doubled and has subsequently decreased to 101 ± 8 mg/m^3 in 2013, near the 2004 baseline value. Mean annual total zooplankton biomass in Northeast Lake has gradually increased between 2008 and 2013, with the exception of a decline in 2012. In Lake 13, mean annual total zooplankton biomass decreased from 2012 to 2013. It is possible that this was a reflection of samples being collected only in August 2012. Despite differences in temporal trends, the 2013 mean annual total zooplankton biomass values were similar among the northwest arm of Snap Lake (101 ± 8 mg/$m^{3)}$, Northeast Lake (90 ± 10 mg/m^3), and Lake 13 (100 ± 9 mg/m^3), with the main basin of Snap Lake being lower (75 ± 4 mg/m^3).

Figure 5-18 Temporal Trends in Mean Annual Zooplankton Biomass in the Main Basin and Northwest Arm of Snap Lake, Northeast Lake, and Lake 13.



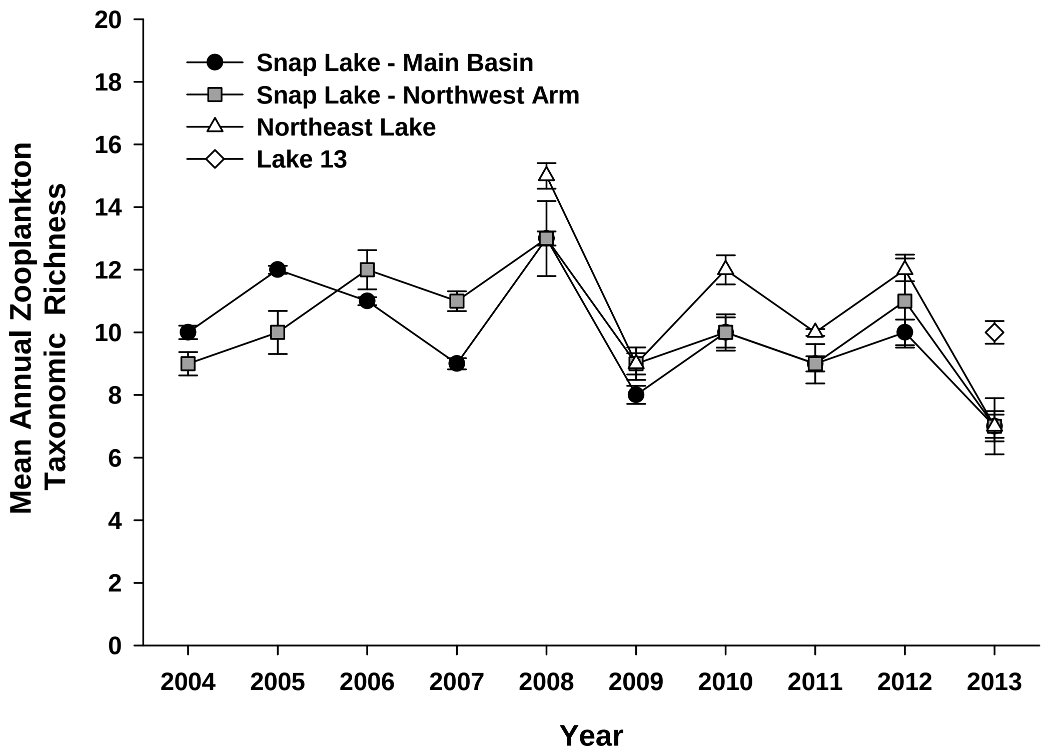
Notes: Error bars represent standard error of the seasonal mean. The 1999 baseline data were not separated into the northwest arm and main basin of Snap Lake (De Beers 2002). Sampling did not occur in Northeast Lake until July 2008 and did not include an August sampling until 2011. Sampling did not occur in Lake 13 until August 2012 and did not include July and September sampling until 2013. The vertical dashed bar represents a break in the time series and change in sampling methods.

mg/m^3 = milligrams per cubic metre.

5.4.4.3 Zooplankton Taxonomic Richness

Mean (± SE) annual total zooplankton taxonomic richness fluctuated over time, but followed similar trends among the main basin and northwest arm of Snap Lake as well as Northeast Lake (Figure 5-19). In 2013, mean annual zooplankton taxonomic richness values in both areas of Snap Lake (7 ± 1 taxa) were below the 2004 baseline value for the main basin (10 ± 1 taxa). Mean annual zooplankton taxonomic richness in Northeast Lake was also lower in 2013 (7 ± 1 taxa) compared to baseline in Snap Lake. In 2013, mean annual zooplankton taxonomic richness in Lake 13 was higher (10 ± 1 taxa) compared with the other areas.

Figure 5-19 Temporal Trends in Mean Annual Zooplankton Taxonomic Richness in the Main Basin and Northwest Arm of Snap Lake, Northeast Lake, and Lake 13.



Note: Error bars represent standard error of the mean. Sampling did not occur in Northeast Lake until July 2008 and did not include an August sampling until 2011. Sampling did not occur in Lake 13 until August 2012 and did not include July and September sampling until 2013.

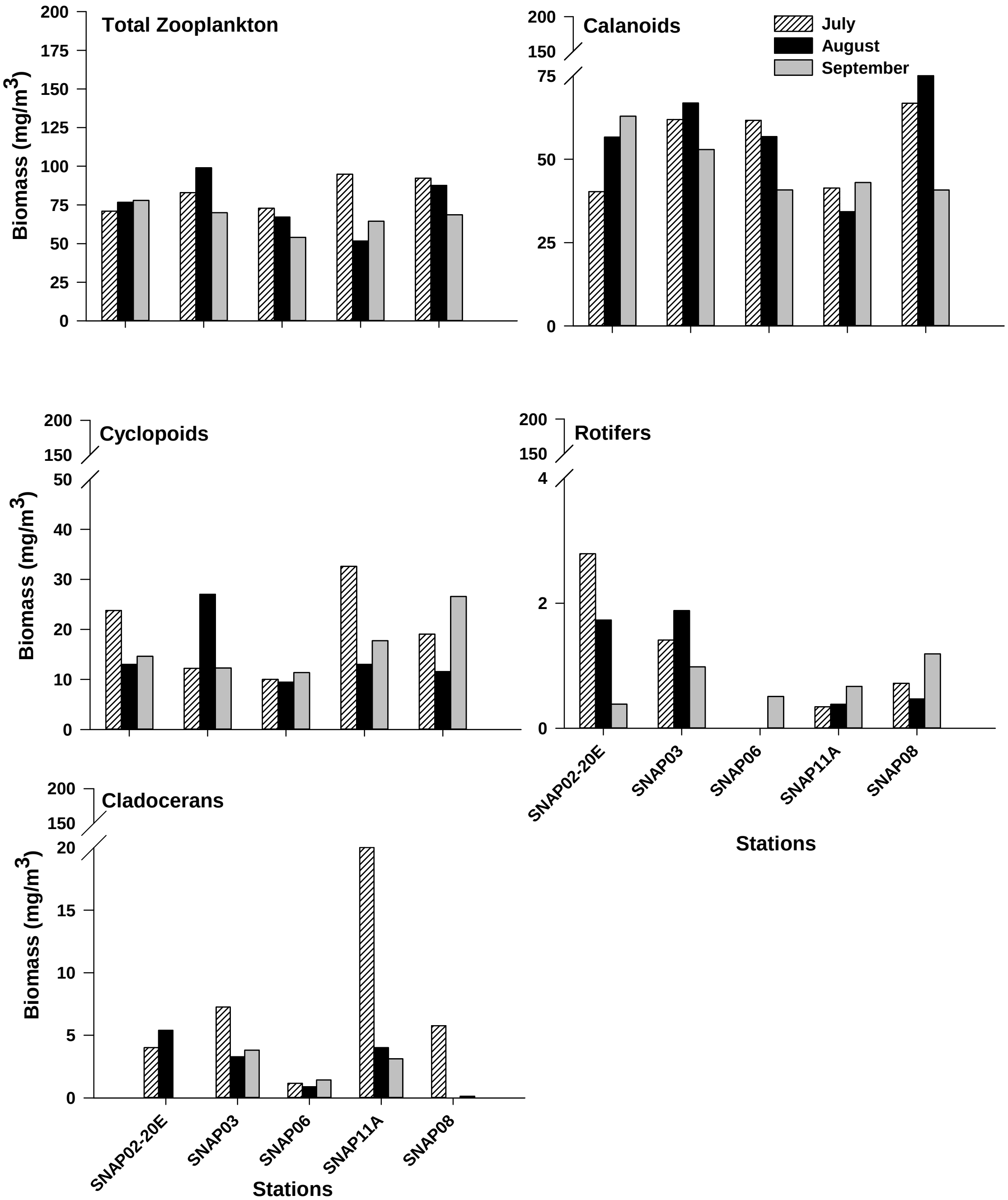
5.4.4.4 2013 Seasonal and Spatial Trends

In general, total zooplankton biomass peaked in July or August in Snap Lake, Northeast Lake, and Lake 13 (Figures 5-20, 5-21, 5-22, and 5-23). Overall, seasonality in total zooplankton biomass was less pronounced in the main basin of Snap Lake compared to the northwest arm. Zooplankton biomass peaked in July in all stations in Northeast Lake and four out of five stations in Lake 13; at Station LK13-01 zooplankton biomass peaked in August.

Seasonal peaks in biomass of the major zooplankton groups varied from group-to-group and from lake-to-lake (Figures 5-20, 5-21, 5-22, and 5-23). Calanoid copepod biomass generally peaked in July or August in all three lakes, with the exception of stations SNAP02-20e, SNAP11A, which had biomass peaks in September, and SNAP23, which had comparable calanoid biomass peaks in July and September. Seasonal peaks in cyclopoid biomass varied between lakes and within lakes and no consistent seasonal patterns were observed. Rotifer biomass generally peaked in July or September, with the exception of LK13-04 and SNAP03, which had biomass peaks in August. Cladoceran biomass generally peaked in July and August in Snap Lake and Northeast Lake. In Lake 13, cladoceran biomass peaked in August at all stations, with the exception of Station LK13-02 in which biomass peaked in September.

No consistent spatial patterns, in relation to proximity to the diffuser, were observed in total zooplankton biomass or the biomass of the major groups in the main basin of Snap Lake in 2013; any spatial variation was likely related to the natural variability in zooplankton community structure (Figure 5-20). In the northwest arm of Snap Lake, total zooplankton and cyclopoid copepods increased with distance from the main basin; however, no spatial patterns were observed in the biomass of the other major groups (Figure 5-21). In general, no consistent spatial patterns were observed in the reference lakes, with the exception of a decrease in rotifer biomass observed in Lake 13 from LK13-02 to LK13-05 (Figures 5-22 and 5-23).

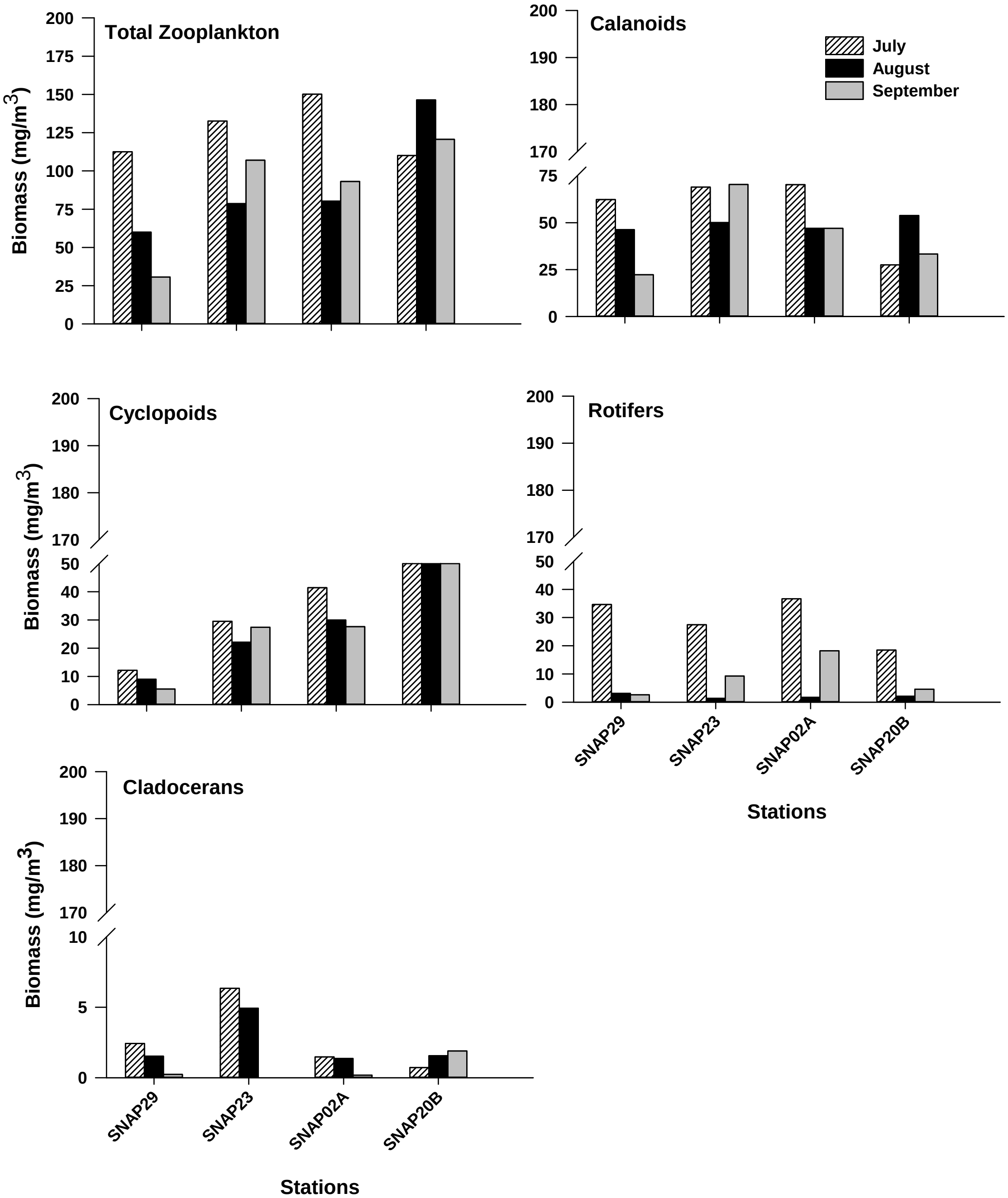
Figure 5-20 Spatial and Seasonal Trends in Biomass of the Major Zooplankton Groups in the Main Basin of Snap Lake, 2013



Note: Stations are arranged from closest to the diffuser (SNAP02-20e) to farthest from the diffuser (SNAP08) in the main basin of Snap Lake.

mg/m^3 = milligrams per cubic metre.

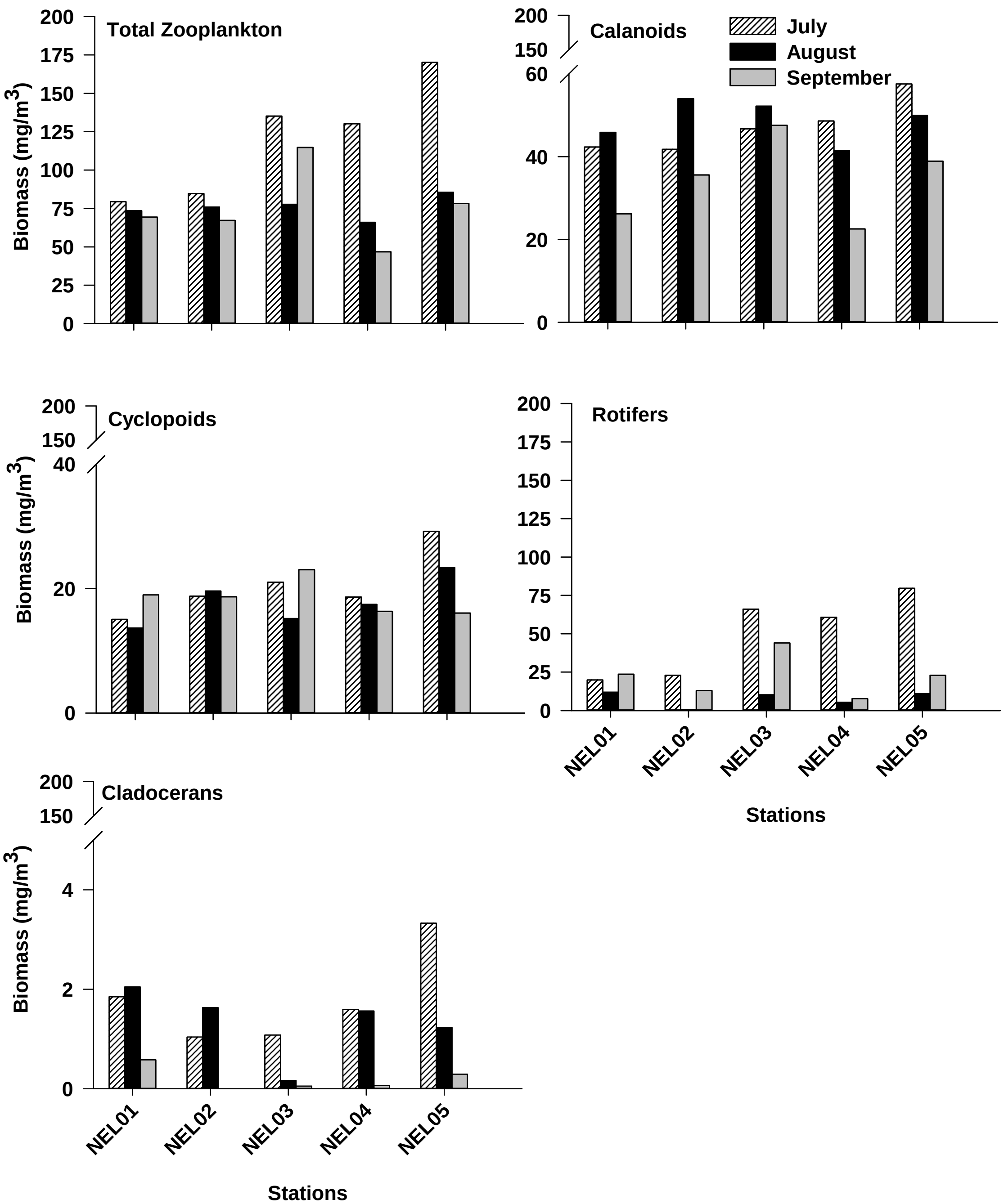
Figure 5-21 Spatial and Seasonal Trends in Biomass of the Major Zooplankton Groups in the Northwest Arm of Snap Lake, 2013



Note: Stations are arranged from closest to the narrows to the main basin (SNAP29) to farthest from the narrows (SNAP20B) in the northwest arm of Snap Lake.

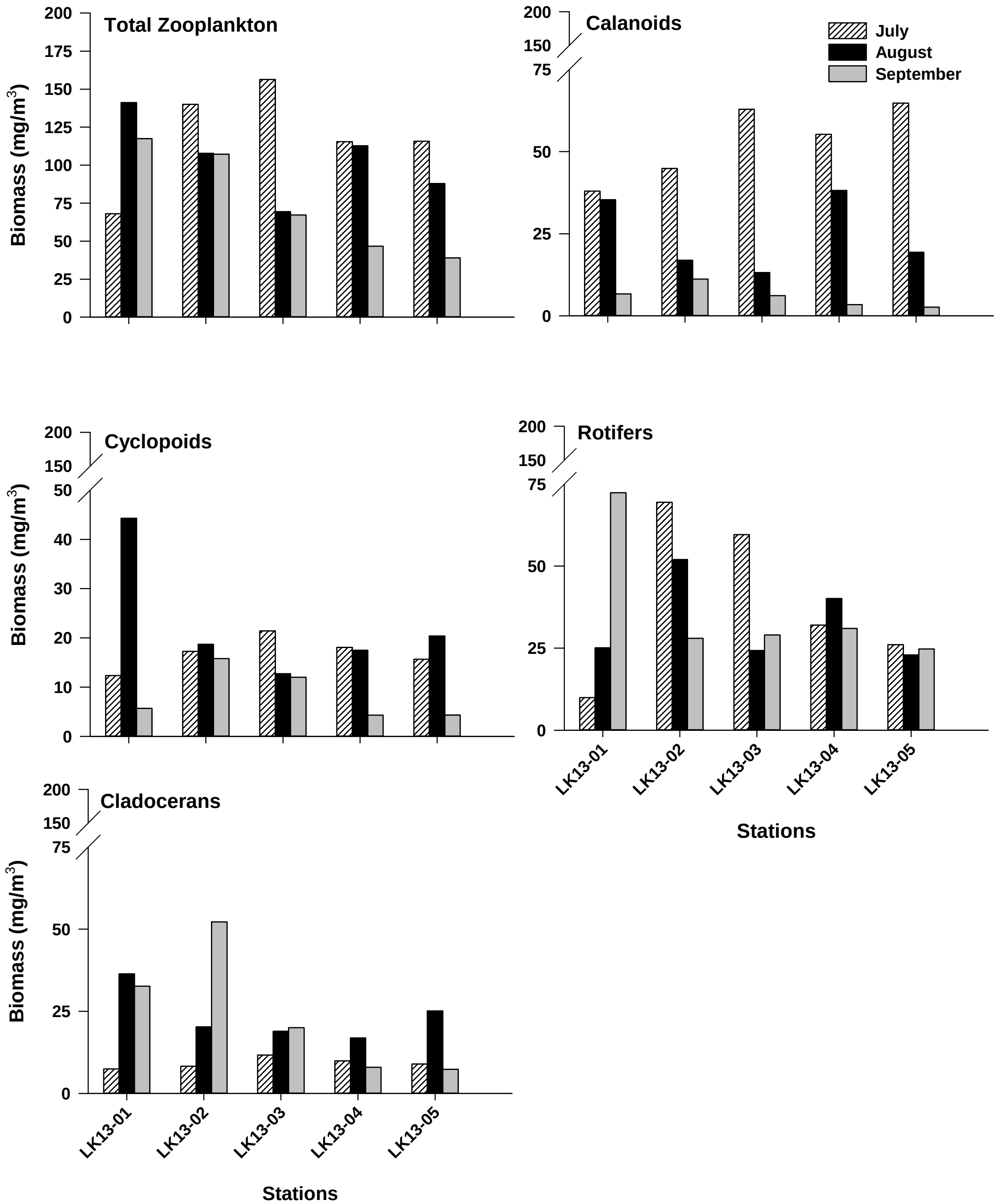
mg/m^3 = milligrams per cubic metre.

Figure 5-22 Spatial and Seasonal Trends in Biomass of the Major Zooplankton Groups in Northeast Lake, 2013



mg/m^3 = milligrams per cubic metre.

Figure 5-23 Spatial and Seasonal Trends in Biomass of the Major Zooplankton Groups in Lake 13, 2013



mg/m^3 = milligrams per cubic metre.

5.4.4.5 Zooplankton Community Composition

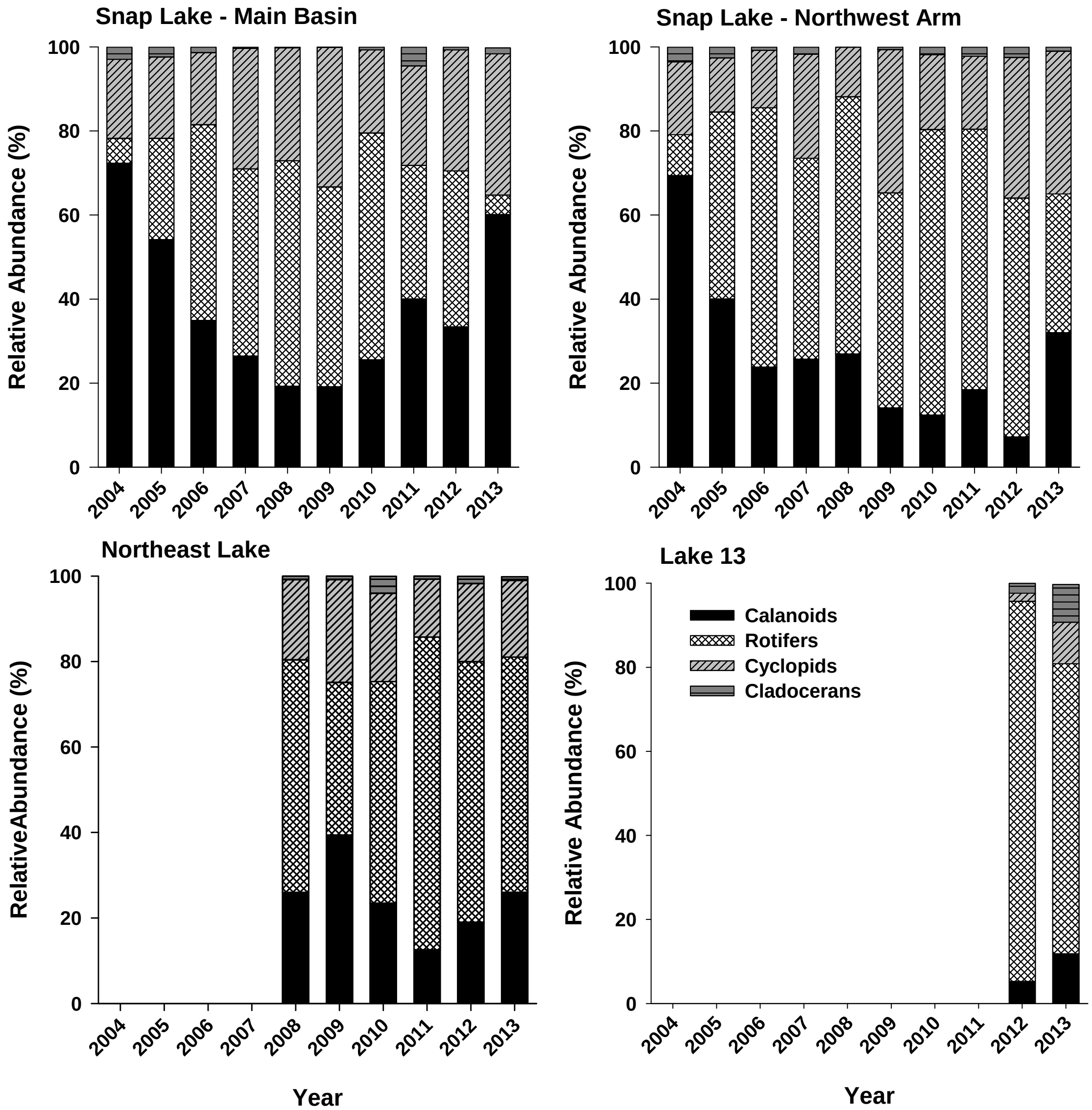
Relative zooplankton abundance in Snap Lake, Northeast Lake, and Lake 13 has varied over time (Figure 5-24). In terms of relative abundance, the main basin of Snap Lake was initially calanoid copepod dominated during baseline (i.e., in 2004 calanoid copepod abundance accounted for approximately 72% of the overall community). From 2004 to 2009, calanoid copepod dominance decreased with increasing dominance of rotifers. From 2009 to 2013, calanoid copepod dominance began to increase, by 2013 accounting for about 60% of the overall community.

Similar to the main basin, zooplankton relative abundance in the northwest arm of Snap Lake was initially dominated by calanoid copepods in 2004 (69%). From 2005 to 2012, calanoid copepod abundance decreased, with concurrent increases in rotifer and cyclopoid copepod abundance (Figure 5-24). In 2013, an increase in calanoid copepod relative abundance was observed in the northwest arm similar to in the main basin of Snap Lake. Overall, zooplankton dominance in the northwest arm of Snap Lake was equally split among the calanoid copepods (31%), cyclopoid copepods (34%), and rotifers (33%).

Between 2008 and 2013, zooplankton relative abundance in Northeast Lake has consistently been dominated by rotifers (35% to 61%). In August 2012, Lake 13 was dominated by rotifers (90%), but in 2013 the relative abundance of rotifers decreased to 69% and calanoid copepod abundance increased from 5% in 2012 to 12% in 2013.

Between 2004 and 2013, relative zooplankton biomass has been consistently dominated by calanoid copepods in Snap Lake (43% to 74%) and Northeast Lake (31% to 65%; Figure 5-25). A decreasing trend in relative biomass of calanoid copepods was observed in both the main basin and northwest arm of Snap Lake between 2004 and 2009. This trend appeared to be reversing in the main basin of Snap Lake from 2009 to 2013; however, it continued in the northwest arm until 2012. The relative proportions of biomass accounted for by rotifers and cyclopoid copepods have varied over time in the northwest arm, with maximum rotifer biomass observed in 2008 and 2010 (39%) and maximum cyclopoid biomass observed in 2012 (44%). In Lake 13, rotifers dominated during the August 2012 sampling period, representing 76% of the community composition based on relative biomass. In 2013, when sampling was completed during all three open-water periods, the community appeared to have a more even distribution among the major groups: rotifers (38%); calanoid copepods (26%); cyclopoid copepods (16%); and, cladocerans (20%).

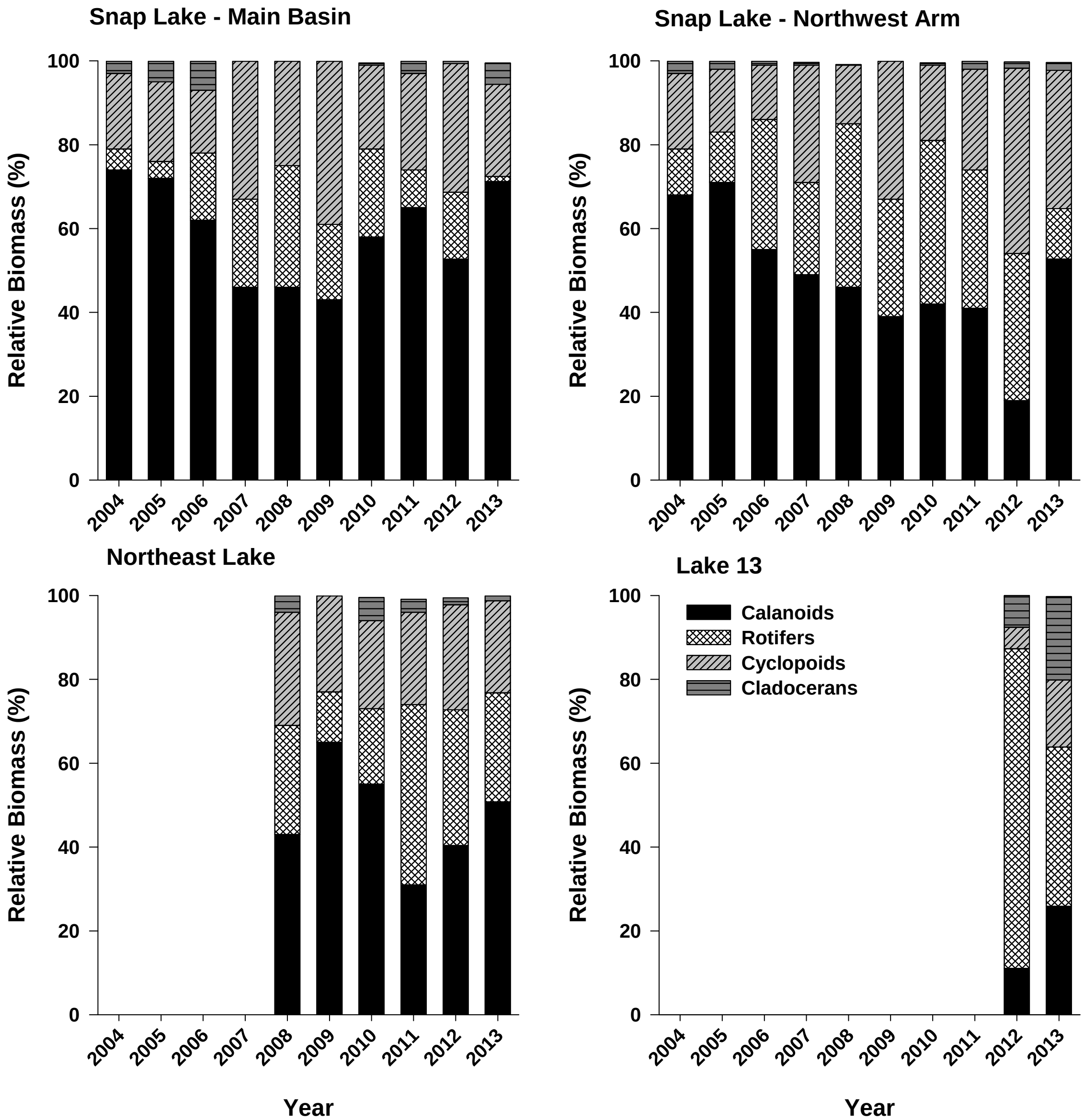
Figure 5-24 Relative Abundance of Zooplankton in the Main Basin and Northwest Arm of Snap Lake, Northeast Lake, and Lake 13



Notes: Sampling did not occur in Northeast Lake until the July 2008 and did not include an August sampling event until 2011. Sampling did not occur in Lake 13 until August 2012 and did not include July and September sampling events until 2013.

% = percent.

Figure 5-25 Relative Biomass of Zooplankton in the Main Basin and Northwest Arm of Snap Lake, Northeast Lake, and Lake 13



Notes: Sampling did not occur in Northeast Lake until the July 2008 and did not include an August sampling event until 2011. Sampling did not occur in Lake 13 until August 2012 and did not include July and September sampling events until 2013.

% = percent.

5.4.5 Toxicity Data Summary

The results of acute and sub-lethal toxicity tests (i.e., *Daphnia magna, Ceriodaphnia dubia*, and *Pseudokirchneriella subcapitata*) completed in 2013 indicate there were no adverse effects related to mortality or reduced reproduction (Section 3; Appendix 3F). As consistently noted previously, growth of *P.subcapitata* was stimulated in all toxicity test samples, with the degree of stimulation increasing at higher sample concentrations.

5.4.6 Edibility Assessment

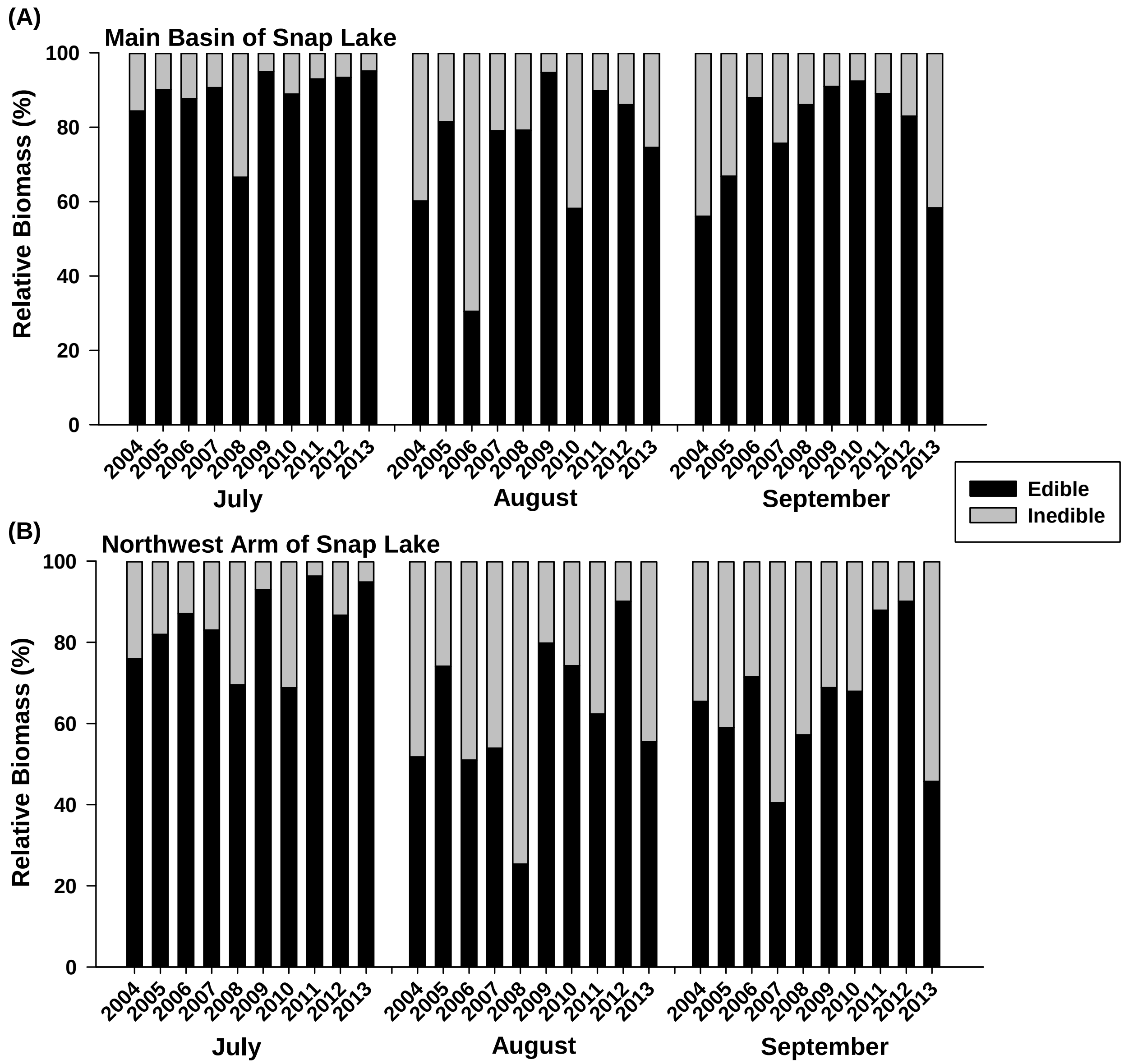
The proportion of edible phytoplankton biomass was high (> 50%) in the main basin of Snap Lake in all three open-water periods, with one exception in August 2006 (Figure 5-26a). In general, the inedible fraction increased as the open-water period progressed within any given year. Greater year-to-year variation was observed in the edible versus inedible biomass data from August and September compared to July. In September there was a steady increase in the edible fraction in the main basin of Snap Lake from 2004 to 2010; however, from 2011 to 2013 a decrease in the edible fraction has been observed.

A higher proportion of inedible phytoplankton was observed in the northwest arm compared to the main basin of Snap Lake in all three open-water periods (Figure 5-26b). Between 2007 and 2013, an increase in edible phytoplankton biomass was observed in September in the northwest arm of Snap Lake; however, a substantial decrease was observed in 2013 (a similar decrease was observed in the main basin of Snap Lake).

Overall, the proportion of edible taxa was lower in Northeast Lake compared to Snap Lake, while the proportion of edible taxa in Lake 13 was similar to that observed in both the main basin and northwest arm of Snap Lake (Figure 5-27). Seasonal variation in the two reference lakes was similar to Snap Lake, with an increase in the inedible fraction occurring as the open-water period progressed each year. In Northeast Lake, a similar increasing trend in the edible fraction from 2008 to 2012 and a decrease in 2013 in September was observed.

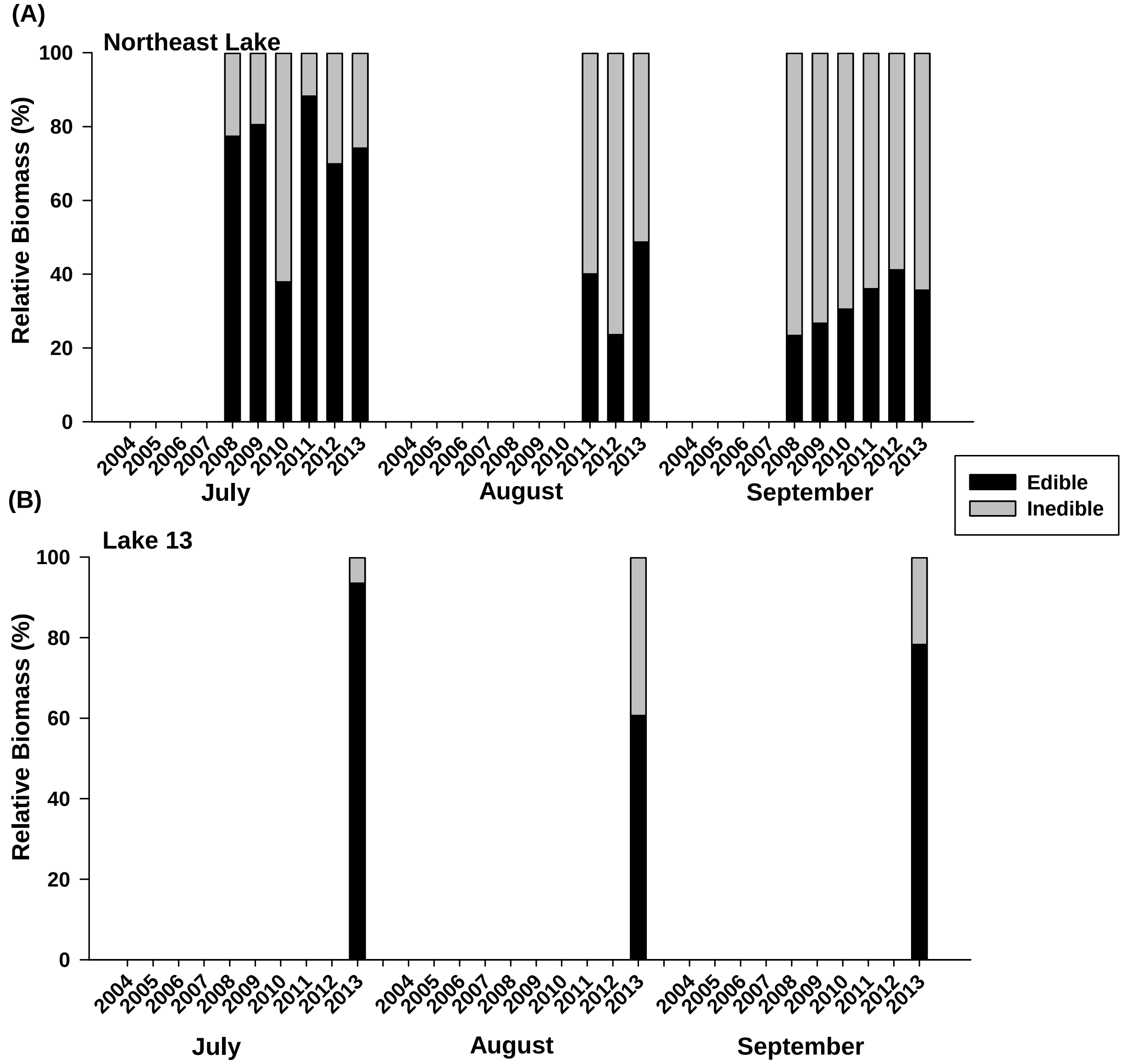
In the main basin of Snap Lake, mean annual edible phytoplankton biomass followed the same increasing trend as mean annual total zooplankton biomass from 2004 to 2007 (Figure 5-28). As grazing pressure declined, as indicated by a decrease in total zooplankton biomass in 2008 and 2009, the edible fraction of the phytoplankton increased. When zooplankton biomass rebounded in 2010 and 2011, a decline in edible phytoplankton biomass was observed, following the classic predator-prey model (Sommer 1996). No clear trend was evident in the inedible fraction of the phytoplankton biomass except in 2008 and 2009, when zooplankton biomass decreased and grazing pressure lessened, the inedible biomass also decreased. No clear relationships were evident between zooplankton biomass and the edible and inedible fractions of the phytoplankton biomass in the northwest arm of Snap Lake, Northeast Lake, or Lake 13 (Figures 5-29 to 5-31).

Figure 5-26 Relative Proportions of Edible and Inedible Phytoplankton Biomass in the Main Basin and Northwest Arm of Snap Lake



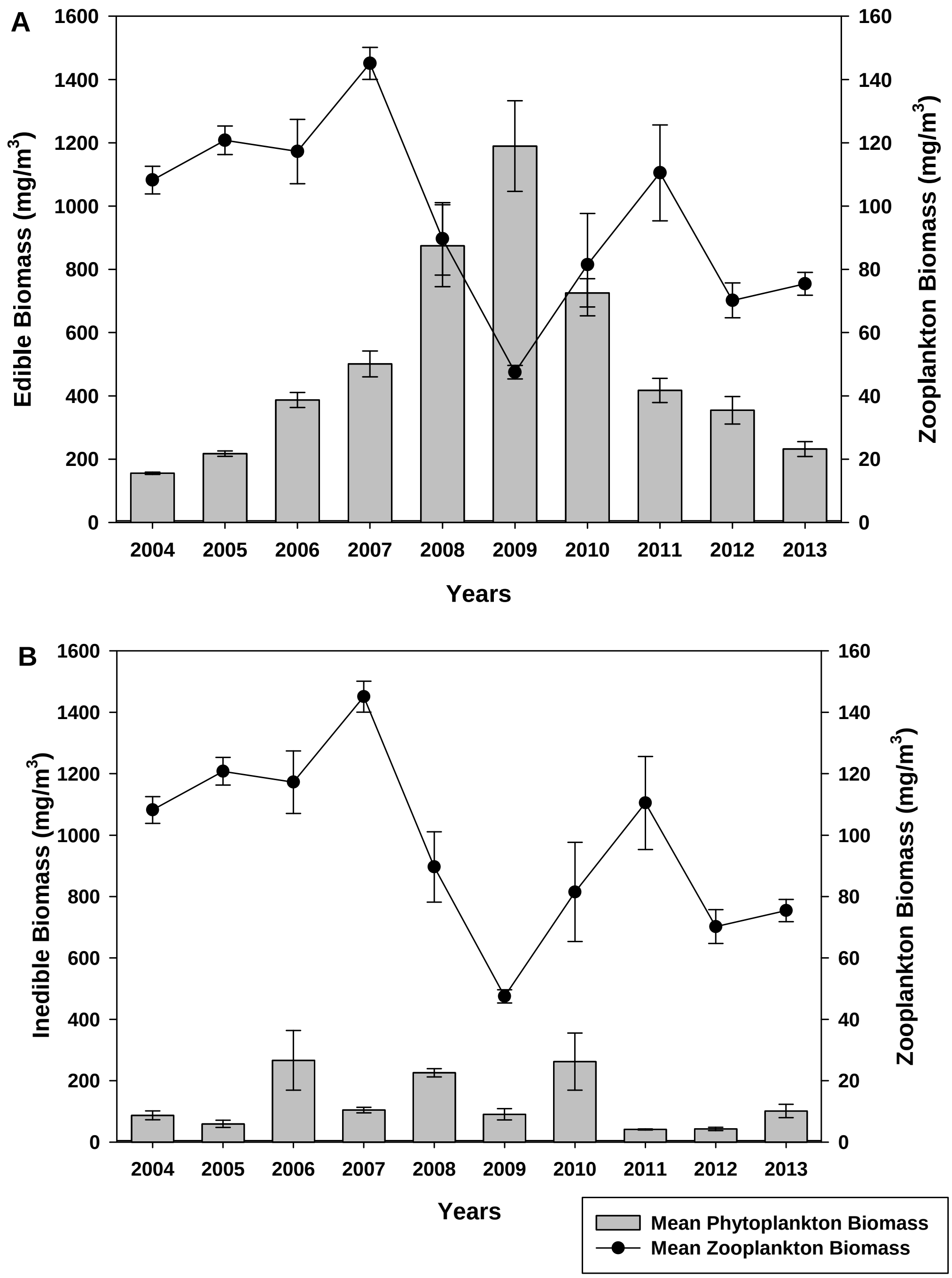
% = percent.

Figure 5-27 Relative Proportions of Edible and Inedible Phytoplankton Biomass in Northeast Lake and Lake 13



% = percent.

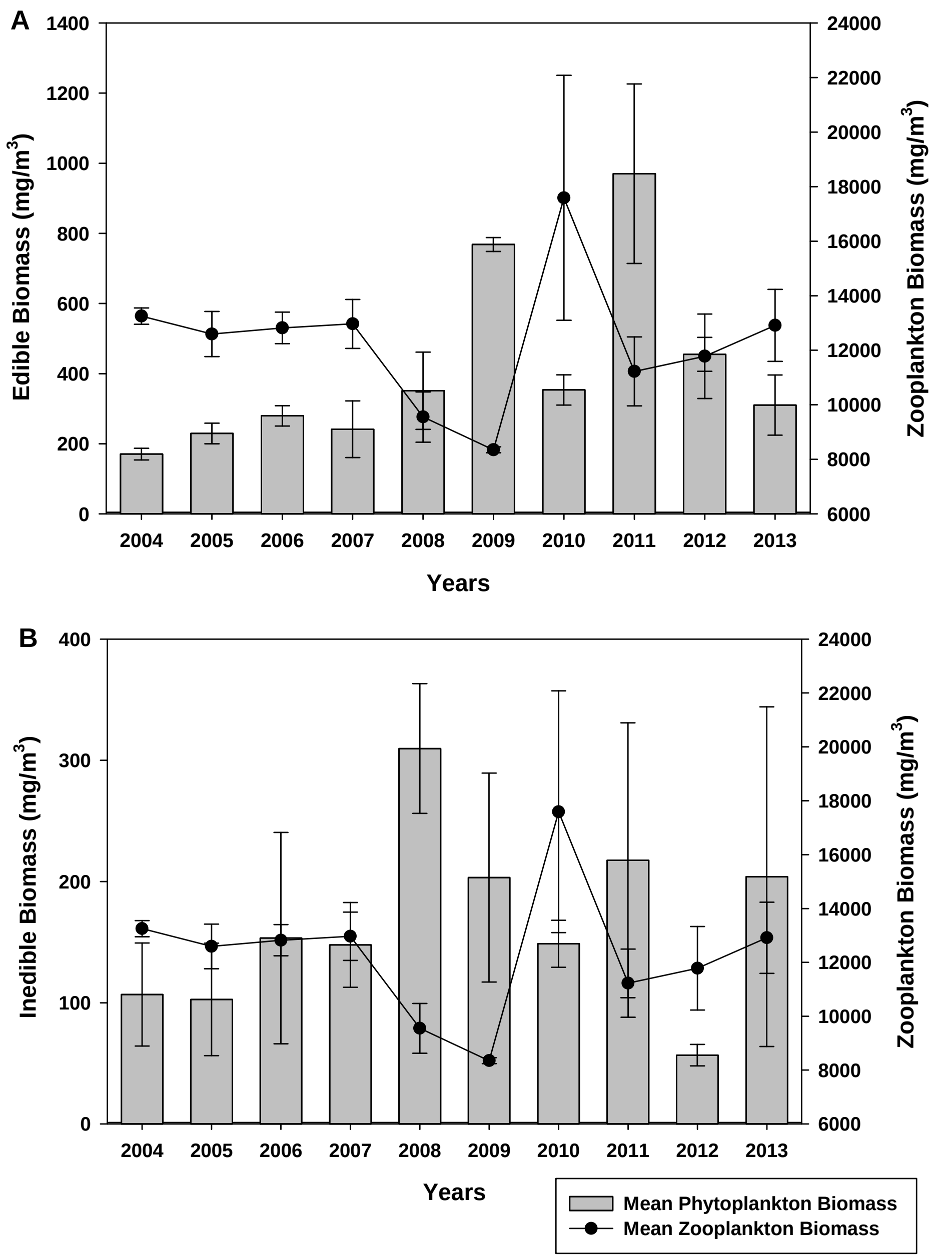
Figure 5-28 Mean Annual Edible (A) and Inedible (B) Phytoplankton Biomass and Mean Annual Zooplankton Biomass in the Main Basin of Snap Lake



Notes: Error bars represent standard error of the mean. Sampling did not occur in Northeast Lake until July 2008 and did not include an August sampling until 2011. Sampling did not occur in Lake 13 until August 2012 and did not include July and September sampling until 2013.

mg/m^3 = milligrams per cubic metre.

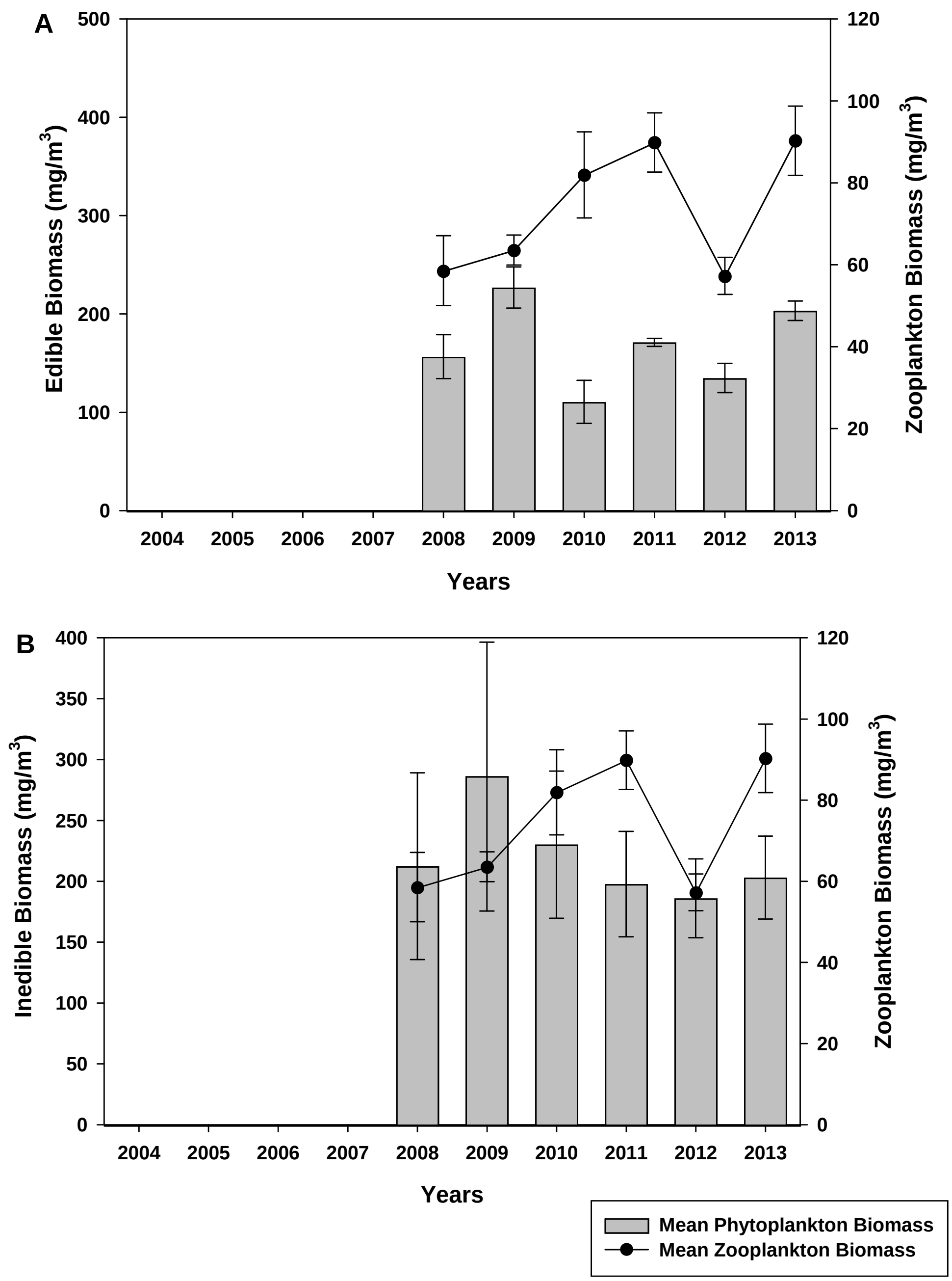
Figure 5-29 Mean Annual Edible (A) and Inedible (B) Phytoplankton Biomass and Mean Annual Zooplankton Biomass in the Northwest Arm of Snap Lake



Notes: Error bars represent standard error of the mean. Sampling did not occur in Northeast Lake until July 2008 and did not include an August sampling until 2011. Sampling did not occur in Lake 13 until August 2012 and did not include July and September sampling until 2013.

mg/m^3 = milligrams per cubic metre.

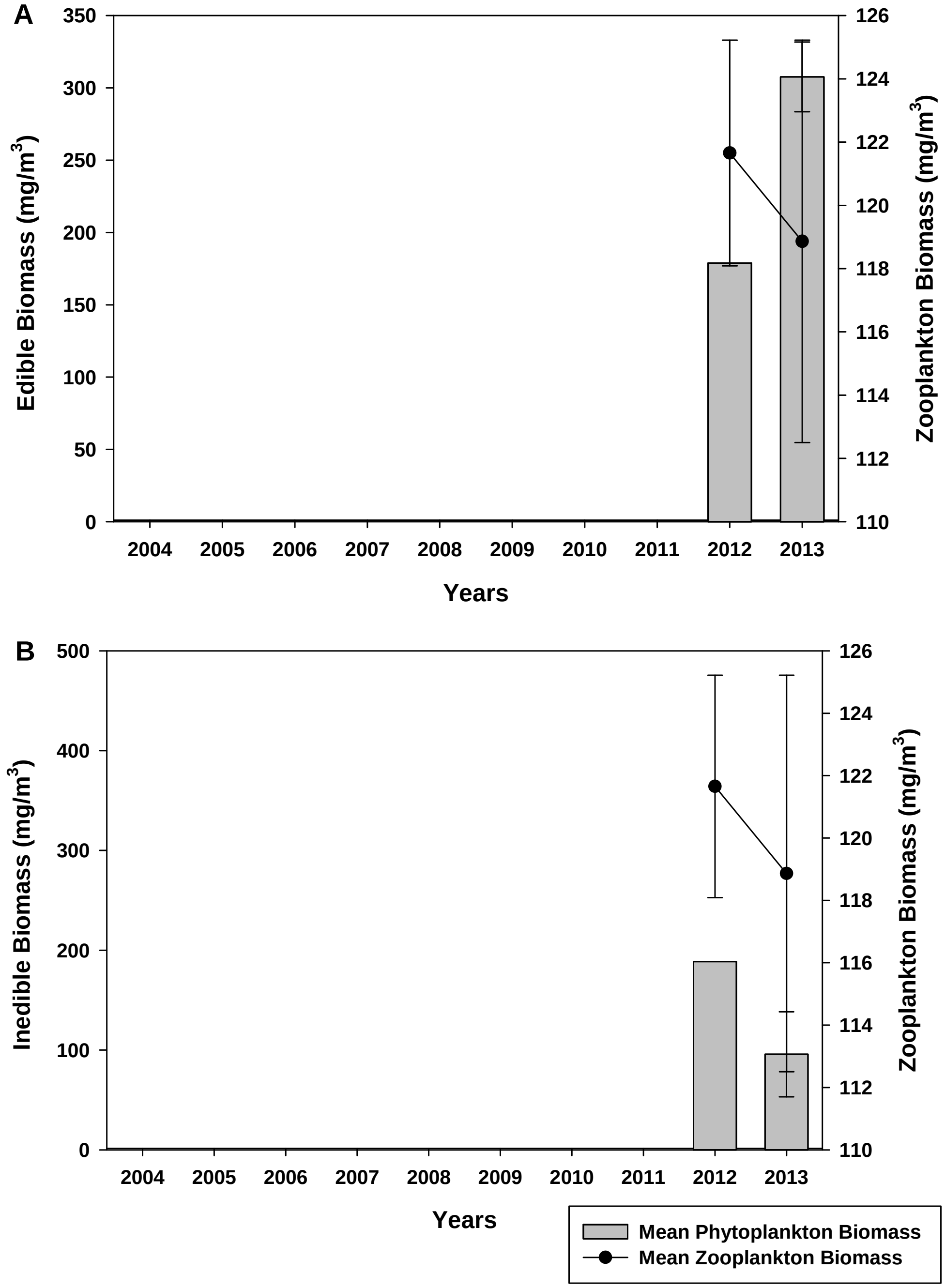
Figure 5-30 Annual Edible (A) and Inedible (B) Phytoplankton Biomass and Mean Annual Zooplankton Biomass in Northeast Lake



Notes: Error bars represent standard error of the mean. Sampling did not occur in Northeast Lake until July 2008 and did not include an August sampling until 2011. Sampling did not occur in Lake 13 until August 2012 and did not include July and September sampling until 2013.

mg/m^3 = milligrams per cubic metre.

Figure 5-31 Mean Annual Edible (A) and Inedible (B) Phytoplankton Biomass and Mean Annual Zooplankton Biomass Lake 13



Notes: Error bars represent standard error of the mean. Sampling did not occur in Northeast Lake until July 2008 and did not include an August sampling until 2011. Sampling did not occur in Lake 13 until August 2012 and did not include July and September sampling until 2013.

mg/m^3 = milligrams per cubic metre.

5.4.7 Environmental Variable Assessment

The local climate at Snap Lake in 2013 was similar to that observed in previous years (Section 2.0). There were 231 days of ice-cover and 134 days of open-water, similar to previous years. Air temperature measurements at the Snap Lake meteorological station indicated the air temperature was within the 1971 to 2000 Yellowknife normal range. Surface water elevation in Snap Lake and Northeast Lake did not change substantially between 2012 and 2013 and was similar to that observed in previous years. Similarly, wind speed and direction did not differ from previous years and rainfall in 2013 was within the 1971 to 2000 Yellowknife normal range. The timing of the rainfall pulse differed in 2013 from in previous years; there was a late summer-fall pulse (August) that was higher than the spring (freshet) pulse, which is normally observed. Average solar radiation during the open-water season has increased from 121 watts per square metre (W/m^2) in 2008 to 165 W/m^2 in 2013.

During the 2013 open-water period, surface water temperatures in Snap Lake varied from 10 to 19 degrees Celsius (°C) and generally remained consistent throughout the water column at all stations, with the exception of the deepest stations in Snap Lake and Northeast lake, which decreased in temperature with increasing depth (Section 3.0). Vertical gradients in DO were not evident during the open-water period in Snap Lake, with the exception of the deepest stations in Snap Lake (i.e., Station SNAP 02 20e and SNAP20B). Concentrations of DO in Northeast Lake and Lake 13 generally remained consistent through the water column. In 2013, field pH ranged across lakes from 5.3 to 8.1, consistent with previous years; there was no evidence of spatial patterns in pH relative to proximity to the diffuser within Snap Lake.

Higher mean conductivity values were observed in the main basin (472 microSiemens per square centimetre [µS/cm^2]) followed by the northwest arm (219 µS/cm^2) of Snap Lake versus the comparatively low conductivity observed in the reference lakes in 2013 (22 µS/cm^{2}and 20 µS/cm^{2}for Northeast Lake and Lake 13, respectively; Section 3.0). Spatial variability within Snap Lake, i.e., decreasing with distance from the diffuser into the northwest arm of Snap Lake, was observed in 2013, consistent with previous years (De Beers 2010, 2011, 2012a, 2013).

Temporal trends in concentrations of TN and Si have been observed in the main basin and northwest arm of Snap Lake since 2004, but have not been observed in Northeast Lake or Lake 13. An increasing trend in concentrations of TP over time has not been observed in Snap Lake despite phosphorus loading to Snap Lake from the treated Mine effluent (Section 3.0).

Higher concentrations of TN and Si were observed in the main basin of Snap Lake, followed by the northwest arm of Snap Lake and then comparatively low concentrations were observed in the reference lakes in 2013 (Section 3.0). Spatial variability within Snap Lake, i.e., decreasing with increasing distance from the diffuser, was observed in concentrations of TN and Si. Clear spatial patterns in TP were not evident, consistent with previous years (De Beers 2010, 2011, 2012a, 2013).

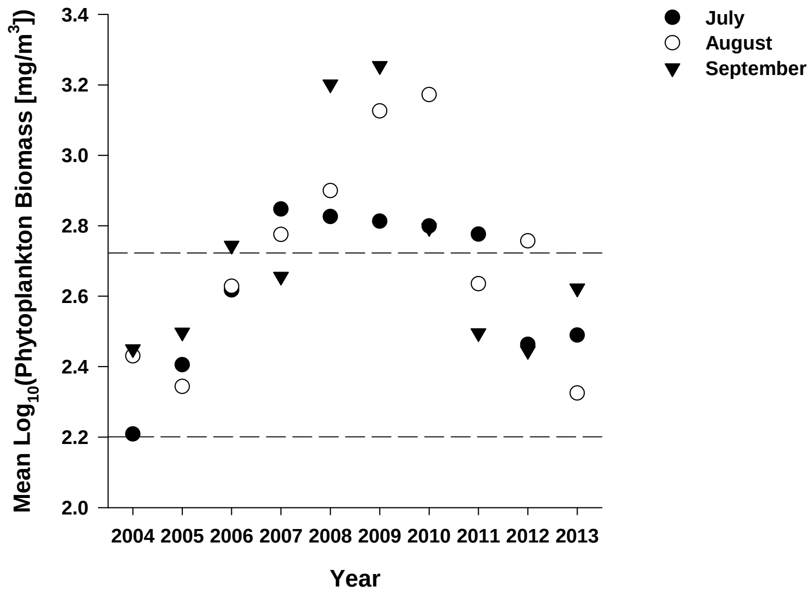
5.4.8 Action Levels Assessment

Plankton communities are inherently variable; therefore, persistent changes need to be observed before action is taken. Changes in the plankton community are considered ecologically important or persistent if they are maintained for three or more years (De Beers 2012b, 2014). A change is documented if differences in trends are observed between Snap Lake and the reference lakes, or if current indicators of change are outside the normal range, as outlined in Section 5.2.8 and Appendix 5B.

For toxicological impairment, persistent declines (i.e., greater than 3 years) beyond the normal range in total phytoplankton biomass or cladoceran abundance or biomass are the indicators of a low Action Level. Neither phytoplankton biomass nor cladoceran abundance or biomass are indicating trends towards toxicological changes. Although there was a historic decrease in cladoceran abundance and biomass, both variables have been increasing since 2009 (Figures 5-32 and 5-33). None of these variable were below the respective normal ranges and no consistent changes were observed, indicating that the plankton communities in Snap Lake are showing negligible toxicological effects.

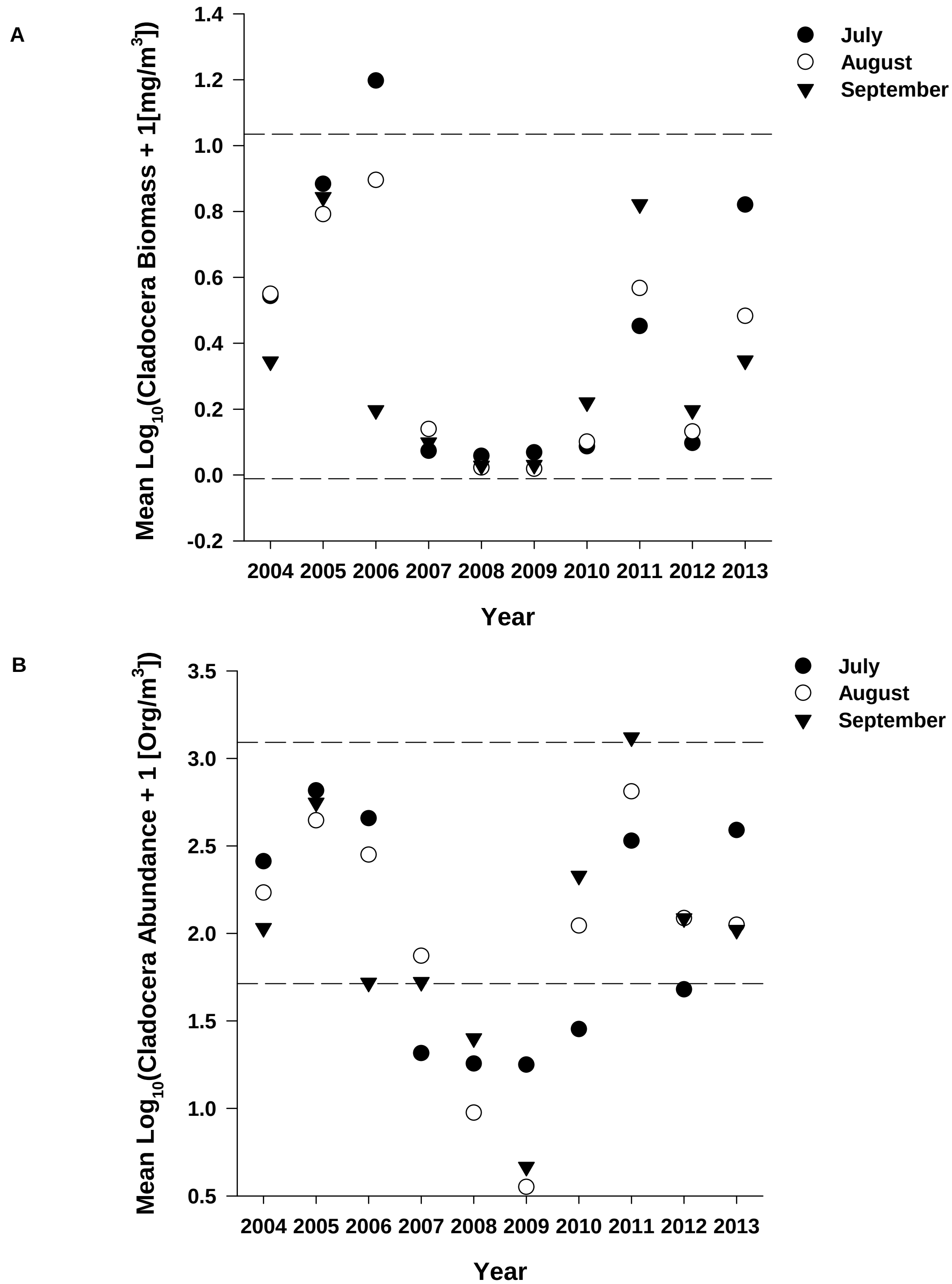
For nutrient enrichment, a persistent increase (i.e., greater than 3 years) beyond the normal range in total phytoplankton and zooplankton biomass in the main basin and a minor shift in phytoplankton or zooplankton community composition are indicators of a low Action Level. Although minor shifts in phytoplankton and zooplankton community composition have been observed (Figure 5-16 and 5-25, respectively), there have been no persistent increases beyond the normal range in either phytoplankton or zooplankton biomass in the main basin of Snap Lake (Figure 5-32 and 5-34, respectively). At this time, the plankton communities in Snap Lake indicate negligible nutrient enrichment effects.

Figure 5-32 Mean Log_{10}(Phytoplankton Biomass) in the Main Basin of Snap Lake, 2004 to 2013



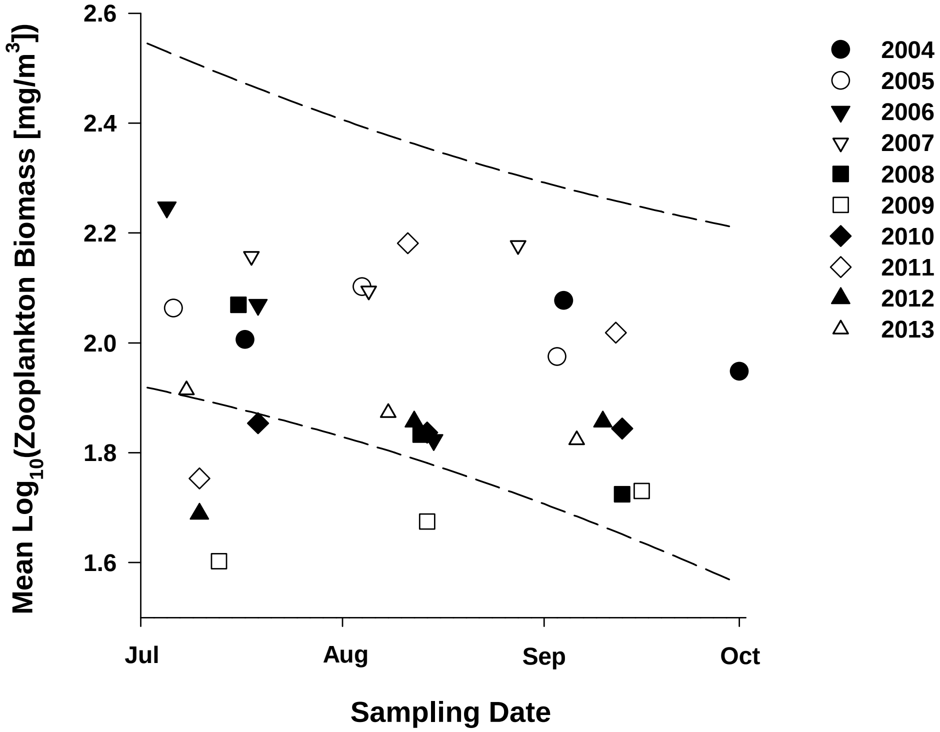
Note: Dashed lines represent the normal range of the unaffected Snap Lake station data from 2004 to 2007.
mg/m^3 = milligrams per cubic metre.

Figure 5-33 Mean Log_{10} (Cladocera Biomass + 1) (A) and Log_{10} (Cladocera Abundance + 1) (B) in the Main Basin of Snap Lake, 2004 to 2013



Note: Dashed lines represent the normal range of the unaffected Snap Lake station data from 2004 to 2007.
mg/m^3 = milligrams per cubic metre; org/m^3 = organisms per cubic metre.

Figure 5-34 Mean Log_{10} (Zooplankton Biomass) in the Main Basin of Snap Lake, 2004 to 2013



Note: Dashed lines represent the normal range of the unaffected Snap Lake station data from 2004 to 2007.
mg/m^3 = milligrams per cubic metre.

5.5 Discussion

5.5.1.1 Light Assessment

Phytoplankton require light for photosynthesis and growth (i.e., increased biomass). The depth of light penetration, and productivity, are determined by the optical properties of the water. A change in water quality is one of the most common factors altering the optical properties and light transmission into water (Kirk 1994). Water quality is determined by alterations in variables such as TDS, TSS, and nutrients. Excess N and P can promote the growth of algal blooms, which can decrease light penetration into the water column. Elevated concentrations of TSS can also reduce the amount of light passing through the water, resulting in a decrease in photosynthesis. Dissolved substances can affect water color, thereby increasing opalescence, and decreasing water column light penetration (Kirk 1994). Conductivity, TDS, and nutrients have increased in Snap Lake since the start of Mine operations (Section 3). These changes can cause changes in light intensity and may be partially contributing to the trends and patterns observed in phytoplankton biomass in Snap Lake.

In 2013, irradiance curves for the plankton stations in the main basin and northwest arm of Snap Lake were characteristic of mesotrophic systems, with the exception of the July SNAP08 irradiance curve, which was most similar to that of an oligotrophic lake. Similarly, the irradiance curves in both reference lakes were characteristic of mesotrophic lakes, with the exception of Station NEL02 in Northeast Lake in September, which had an irradiance curve most similar to an oligotrophic lake. These irradiance curves are consistent with the trophic classification of Snap Lake as a meso-oligotrophic lake as outlined in the EAR (De Beers 2002).

From 2004 to 2013, the top 6 m of the water column was used as the estimated euphotic zone (i.e., greater than 1% surface irradiance) in Snap Lake, Northeast Lake, and Lake 13. This is the zone where light is sufficient for phytoplankton photosynthesis; therefore, this was the zone in which phytoplankton, chlorophyll *a*, depth-integrated nutrients, microcystin, and picoplankton were collected (Section 5.2.4.2). In 2013, light meter measurements indicated that the euphotic zones in Snap Lake, Northeast Lake, and Lake 13 generally extended to the bottom depths, with the exception of the deep stations in the main basin (SNAP02-20e) and the northwest arm (SNAP20B) of Snap Lake. Overall, the main basin of Snap Lake had the deepest mean euphotic zone depth (12 m), followed by Northeast Lake (11 m), Lake 13 (9 m), and the northwest arm of Snap Lake (9 m), all of which are deeper than the estimated zone of 6 m. Irradiance at the 6 m depth was well above 1%, and ranged from 4% in September to 25% of surface irradiance in July (Appendix 5A; Addendum 5A-1). This indicates that a small portion of the phytoplankton community may have been overlooked based on sampling protocols followed since baseline (i.e., 2004). Changes to the sampling protocols are not recommended, and sampling of the top 6 m of the water column should continue for all depth-integrated sampling to remain consistent with previous year's data; however, it should be noted that a slight under-representation of the phytoplankton community may have been reported.

Although the Secchi depth estimates of the euphotic zone were consistently lower than those measured with the light meter in 2013, these differences were less than anticipated indicating that the Secchi depth method was a more reliable method of estimating the euphotic depth than predicted. Secchi depth measurements can provide a coarse estimate of euphotic zone depth. However, the light meter data indicate that differences are being observed in the irradiance curves between stations, particularly in the main basin of Snap Lake. These differences cannot be observed with Secchi depth measurements; however, it is recommended that both light meter measurements and Secchi depth measurements are continued to support the plankton component. The Secchi depth measurements allow an alternative to the light meter measurements in case of an equipment malfunction.

5.5.1.2 Chlorophyll *a* and *c* Concentrations

Chlorophyll *a* concentrations in the main basin and northwest arm of Snap Lake varied between 2004 and 2013, with no clear temporal or spatial trends observed in either area. Chlorophyll *a* concentrations increased in the main basin of Snap Lake between 2011 and 2013, Northeast Lake between 2010 and 2013, and Lake 13 between 2012 and 2013. This suggests that environmental conditions, such as increased surface irradiance may be affecting chlorophyll *a* concentrations more than Mine-related conditions. Chlorophyll *a* concentrations in 2013 ranged from 0.24 to 3.11 µg/L; the upper bound of this range has increased beyond the EAR predictions that chlorophyll *a* would gradually increase to a range of 1.5 to 2.3 µg/L over a 20 year period (De Beers 2002). However, the mean annual chlorophyll *a* concentration (1.06 µg/L) still falls within the EAR prediction that the productivity status of Snap Lake would remain unchanged and stay within the range of oligotrophic lakes (0.3 to 4.5 µg/L; Wetzel 2001).

Concentrations of chlorophyll *c* were anticipated to increase with increases in diatom and chrysophyte biomass; however, no clear relationships have been observed in 2012 (De Beers 2013) or 2013. Chlorophyll *c* concentrations in 2013 were below detection limits at many of the stations sampled and were mostly below or within the range of the 2005 values (0.01 to 1.7 µg/L; De Beers 2013), with the exception of elevated duplicate values observed in the northwest arm in July at SNAP29 (5.12 µg/L and 4.64 µg/L).

5.5.1.3 Phytoplankton Community

Mean annual total phytoplankton abundance has increased since baseline (i.e., 2004) in the main basin and northwest arm of Snap Lake, but has remained relatively consistent since 2009. In Northeast Lake, mean annual total phytoplankton abundance has remained consistently since 2008 below values observed in Snap Lake. Similarly, although mean annual phytoplankton taxonomic richness, based on a genus-level assessment, varied over time in the three study lakes, taxonomic richness values were higher in Northeast Lake and Lake 13 compared to the main basin and northwest arm of Snap Lake.

Biomass is an important endpoint relative to predation and overall ecological importance; therefore, it is often viewed as a more critical phytoplankton endpoint to monitor than abundance (Sommer 1996). In terms of phytoplankton biomass, the phytoplankton community in the main basin of Snap Lake has undergone two community shifts: one from a chrysophyte-cyanobacteria dominated community to a diatom-dominated community; and, back to a chrysophtye-cyanobacteria dominated community. These changes in community structure followed a close trend to changes in overall phytoplankton biomass. An increase in overall phytoplankton biomass from 2004 to 2009 was observed, related to an increase in diatom biomass. Subsequently, a decrease in overall phytoplankton biomass from 2009 to 2013 was observed, related to a decrease in diatom biomass. The main basin is now fully mixed, with no consistent spatial patterns in phytoplankton variables in relation to proximity to the diffuser. Overall, the main basin appears to be returning to a more stable state, similar to baseline conditions; however, biomass was still 1.5 times greater than the baseline value in 2013.

In the northwest arm of Snap Lake, biomass was lower than in the main basin until 2011. The phytoplankton community in the northwest arm of Snap Lake has consistently been chrysophyte-dominated with sub-dominance of diatoms gradually increasing over time. The changes that were observed in the main basin from 2004 to 2009 (i.e., increased overall biomass and increased diatom dominance) appear to now be occurring in the northwest arm of Snap Lake. The lag in response to Mine-related activities is consistent with the slow spread of treated effluent into the northwest arm of Snap Lake; changes to the phytoplankton community in the northwest arm are just now being observed.

There has been little change in phytoplankton biomass in Northeast Lake. In 2013, phytoplankton biomass in Lake 13 was similar to biomass observed in Northeast Lake, but higher than biomass observed in the main basin and lower than observed in the northwest arm of Snap Lake. Phytoplankton biomass in Northeast Lake has been cyanobacteria-chrysophyte co-dominated since sampling began in 2008. Phytoplankton biomass in Lake 13 differed from both Northeast Lake and Snap Lake, and was chrysophyte-dominated with sub-dominance by chlorophytes, diatoms, and dinoflagellates. Overall, phytoplankton biomass generally peaked in September in all three lakes and seasonal peaks in biomass of the major phytoplankton groups varied from group-to-group and from lake-to-lake.

Generally, an increase in cyanobacteria biomass, rather than diatom biomass, is expected with nutrient enrichment (Wehr and Sheath 2003). Cyanobacteria respond to increases in available P that occur with nutrient enrichment, particularly when the system becomes more N-limited; most species of cyanobacteria are capable of fixing atmospheric nitrogen (N_2). However, in Snap Lake there is a substantial N-load from the treated Mine effluent, which has caused increased P-limitation. Therefore, the N_2-fixing cyanobacteria do not have a competitive advantage over other groups of phytoplankton in the main basin of Snap Lake and cyanobacteria would not be expected to dominate.

Overall, cyanobacteria biomass has fluctuated in Snap Lake over the years and differed between the main basin and the northwest arm (De Beers 2012b). Cyanobacteria biomass increased from 2012 to 2013 in the main basin and northwest arm of Snap Lake; this increase was caused by an increase in a colonial taxa: *Aphanocapsa delicatissima* (Appendix 5C, Table 5C-6). *A. delicatissima* is both a bloom-forming and toxin-forming taxa (Wehr and Sheath 2003); however, conditions were not sufficient for blooms to form in Snap Lake and, in addition, microcystin concentrations in all areas were near or below detect in 2013 (Section 3.0).

Diatoms require high Si concentrations for cell wall development and Si is often the nutrient limiting diatom growth in many lakes (Wehr and Sheath 2003); however the sustained inputs of Si from treated effluent provides surplus Si for diatom growth in Snap Lake. The diatom community in the main basin of Snap Lake may have undergone an early increase in biomass as a result of the newly available Si and favorable conditions for diatom growth; however, the sustained Si being provided to the system may no longer be favorable for the opportunistic diatoms to outcompete other phytoplankton groups in the main basin and this is reflected in a decrease in diatom dominance in the main basin. In the northwest arm of Snap Lake, the gradual increase in diatoms is likely related to the increased spread of treated effluent water, and thus increased Si, into the northwest arm of Snap Lake.

Chrysophytes are known to frequently dominate or co-dominate phytoplankton biomass and abundance in clear oligotrophic lakes with low alkalinity (0 to 60 mg/L $CaCO_3$), conductivity (less than 50 µS/cm), and nutrient concentrations (Wehr and Sheath 2003), as is seen in the reference lakes. The decrease in chrysophyte dominance in Snap Lake between 2006 and 2012 may have been associated with changes in water hardness and adjustments to increases in conductivity, silica, and alkalinity which are linked to Mine operations (De Beers 2012b).

Toxicity testing demonstrated stimulation in algal growth (i.e., *Pseudokirchneriella subcapitata*) rather than inhibition, with the degree of stimulation increasing at higher sample concentrations of Snap Lake water (Appendix 3F). These toxicity test results are in agreement with what is being observed in the phytoplankton community, i.e., despite a decrease in phytoplankton biomass, a negligible enrichment effect in Snap Lake rather than toxicological impairment affect is apparent.

The EAR (De Beers 2002) predicted a slight increase in phytoplankton abundance and biomass, as well as a minor shift in phytoplankton community structure, where the relative proportion of various species may change but no loss of species and no major shift in keystone species would occur. The 2013 results fall within the EAR predictions for abundance and biomass, showing no substantial changes in 2013. However, a minor shift in community structure continues to be observed. Generally, the phytoplankton community in the main basin of Snap Lake is showing signs of decreasing variability and a return to conditions similar to those observed at baseline.

5.5.1.4 Zooplankton Community

Mean annual total zooplankton abundance and taxonomic richness varied over time in Snap Lake, Northeast Lake, and Lake 13. No clear temporal trend was observed in mean annual total zooplankton abundance in the main basin or the northwest arm of Snap Lake. Mean annual total zooplankton abundance in the main basin has generally remained near the 2004 baseline value, with some fluctuations occurring over time. The northwest arm of Snap Lake mean annual total zooplankton abundance has been consistently higher than in the main basin, whereas, mean annual total zooplankton abundance in Northeast Lake has been similar to that observed in the main basin of Snap Lake since 2008. An overall decreasing trend was observed in taxonomic richness in Snap Lake and Northeast Lake; by 2013 all areas were below baseline values observed in the main basin of Snap Lake in 2004. No changes were observed between 2012 and 2013 in Lake 13. Changes in abundance have occurred in all three lakes, which suggest that there is not a strong Mine-related influence on zooplankton abundance;

similarly, changes in taxonomic richness were observed in both Snap Lake and Northeast Lake, suggesting regional factors may be affecting zooplankton taxonomic composition, rather than Mine-related factors.

Mean annual zooplankton biomass decreased over time in Snap Lake and from 2009 to 2013 zooplankton biomass has remained below baseline values. In the northwest arm of Snap Lake zooplankton biomass decreased from 2006 to 2009, followed by an increase in zooplankton biomass in 2010. After 2010, zooplankton biomass in the northwest arm continued to decrease to near baseline values. Conversely, zooplankton biomass in Northeast Lake has increased since 2008. Overall, zooplankton biomass was similar among areas in 2013. Seasonal peaks in total zooplankton biomass were observed in July or August in all three areas. No consistent spatial patterns, in relation to proximity to the diffuser, were observed in total zooplankton biomass or the biomass of the major groups in the main basin of Snap Lake; however, in the northwest arm of Snap Lake, cyclopoid copepods increased with increasing distance through the northwest arm from the main basin.

In terms of zooplankton biomass, the zooplankton community in Snap Lake and Northeast Lake has been consistently dominated by calanoid copepods; however, in the main basin of Snap Lake the community has undergone a community shift from a calanoid copepod dominated community during baseline to a cyclopoid-calanoid copepod co-dominated community from 2007 to 2012 back to a mainly calanoid copepod dominated community in 2013. In the northwest arm of Snap Lake, calanoid copepods decreased in dominance from 2004 to 2009, and increased in dominance from 2009 to 2013, following the trend observed in the main basin of Snap Lake. Zooplankton community composition in Northeast Lake was similar to that in Snap Lake; however, composition in Lake 13 differed, i.e., rotifers dominated in 2012 but a more even distribution among the major groups was observed in 2013. The zooplankton community in Northeast Lake is more similar than the Lake 13 community to the zooplankton community observed during baseline in Snap Lake; however, Lake 13 does provide an understanding of how variable these communities can be from one system to another under natural conditions.

The EAR (De Beers 2002) predicted a gradual lake-wide increase in TDS. The predicted magnitude of this effect on plankton was classified as negligible. However, an increase in the maximum predicted calcium concentrations (110 mg/L) in Snap Lake was predicted to have a low magnitude effect on zooplankton, specifically causing an increase in cladoceran abundance and biomass. An increase in calcium concentrations was observed in the main basin of Snap Lake (De Beers 2013). However, from 2004 to 2013, relative percent cladoceran biomass has remained low in all of the lakes.

Cladoceran biomass in Snap Lake has been low compared to Northeast Lake from 2008 to 2013 and Lake 13 in 2012 and 2013. Cladoceran biomass has been consistently low in Snap Lake; however, from 2007 to 2010 cladoceran biomass was particularly low. From 2010 to 2013 cladoceran biomass in Snap Lake has increased. The primary cladoceran species observed in Snap Lake in 2012 was *Daphnia longerimis*; whereas in 2013 the dominant cladoceran was *Daphnia pulex* (Appendix 5C, Table 5C-9). Although the cause of the temporary decline in cladoceran biomass in Snap Lake is not known, shifts in plankton species composition can occur naturally and can be cyclical over time. If the change in cladoceran biomass was related to Mine activities, it is contrary to what was predicted in the EAR. Because cladoceran biomass in Northeast Lake was also low in 2009, this suggests that a regional phenomenon not associated with the Mine caused the decline in Snap Lake from 2007 to 2010. Overall, cladoceran biomass has increased in both the main basin and northwest arm of Snap Lake in recent years.

In oligotrophic systems, copepods are generally the dominant zooplankton group by abundance (Carney 1990). In Snap Lake, copepods dominated during baseline. Copepod abundance in Northeast Lake has been consistent with abundances observed in Snap Lake from 2008 to 2012. The grazing rate of copepods is lower than cladocerans; as a result, copepods do not have as great an effect on phytoplankton community structure or biomass (Wetzel 2001). In addition, copepods have a much lower per capita filtering rate compared to cladocerans and they excrete faecal pellets rather than dissolved N and P, which cladocerans actively excrete and regenerate in their soluble available forms. This enables phytoplankton productivity and speeds nutrient cycling, and tightens the coupling between phytoplankton and zooplankton. However, increased copepod biomass can lead to a reduced coupling of phytoplankton and zooplankton (Carney 1990). In general, low zooplankton grazing rates would favour edible phytoplankton species. High zooplankton grazing rates favour inedible species because nutrients regenerated from the digestion of edible species supply a continuous nutrient supply for rapid growth of the poorly edible or inedible taxa (Sterner 1989). Planktivorous fish often select for large cladocerans, while larval fish select for copepods and rotifers (Carney 1990). It is possible that heavy grazing pressure from planktivorous fish (i.e., Round Whitefish) may be quickly reducing cladoceran biomass and abundance in Snap Lake and Northeast Lake.

Toxicity testing demonstrated that there were no adverse effects for any of the cladoceran test endpoints, e.g., *Daphnia magna* or *Ceriodaphnia dubia* (Appendix 3F) in Snap Lake in 2013. These tests suggest that the decreases in total zooplankton biomass observed in the zooplankton community since baseline may not be caused by toxicological impairment but rather are simply natural fluctuations in the community.

The EAR (De Beers 2002) predicted a slight increase in zooplankton abundance and biomass, which could lead to a minor increase in fish food and a minor change in zooplankton community structure. It stated that the relative proportion of various species may change, but no loss of species and no major shifts in keystone species were expected. The EAR prediction of negligible increases is consistent with the low magnitude changes that have been observed in zooplankton biomass and abundance in Snap Lake. However, a minor shift in community structure is being observed.

5.5.1.5 Edibility Assessment

The edibility assessment indicates that edible taxa comprise the majority of the phytoplankton communities in all three lakes. This suggests that the integrity of the planktonic ecosystems has not been compromised. However, it is possible that Snap Lake is in a transitional stage and the niche required for inedible phytoplankton to flourish has not yet been created, but may occur with continued nutrient inputs (De Beers 2012b); therefore, continued monitoring is recommended including size class categorization of the data.

5.5.1.6 Action Levels Assessment

For toxicological impairment, no variables were below the normal range and no consistent changes were observed in phytoplankton biomass or cladoceran abundance or biomass indicating that the plankton communities in Snap Lake are showing negligible toxicological effects. For nutrient enrichment, although minor shifts in phytoplankton and zooplankton community composition were observed, there were no persistent increases beyond the normal range in either phytoplankton or zooplankton biomass in the main basin, indicating that the plankton communities in Snap Lake are also showing negligible nutrient enrichment effects.

5.6 Conclusions

5.6.1 Key Question 1: What are the Current Concentrations of Chlorophyll *a* and *c*, and What Do These Concentrations Indicate about the Trophic Status of Snap Lake, Northeast Lake, and Lake 13?

Chlorophyll *a* concentrations in Snap Lake have varied between 2004 and 2013, with no clear temporal trend in either the main basin or northwest arm. Chlorophyll *c* concentrations have not increased in Snap Lake since sampling in 2005. There is no consistent spatial trend in mean chlorophyll *a* and c concentrations between the main basin and northwest arm of Snap Lake. In 2013, chlorophyll *a* concentrations in the main basin and the northwest arm of Snap Lake remained within the range characteristic of oligotrophic lakes (0.30 to 4.5 µg/L; Wetzel 2001).

5.6.2 Key Question 2: What is the Current Status, in Terms of Abundance, Biomass and Composition, of the Phytoplankton Community in Snap Lake, Northeast Lake, and Lake 13, and do these Results Suggest Signs of Mine-Related Nutrient Enrichment or Toxicological Impairment?

Mean annual phytoplankton abundance has increased since baseline (i.e., 2004), and is currently higher than baseline, but has remained relatively consistent from 2007 to 2013. There has been no change over time in mean annual phytoplankton abundance in Northeast Lake. In 2013, mean annual phytoplankton biomass within the main basin of Snap Lake was approximately 1.5 times higher than baseline. Phytoplankton biomass in the main basin of Snap Lake continued to decrease in 2013. Differences between the northwest arm and the main basin of Snap Lake are widening, with phytoplankton biomass in the northwest arm increasing in 2013.

The phytoplankton community composition within the main basin of Snap Lake has shifted from a diatom-chrysophyte co-dominated community in 2012 to a chrysophyte-dominated community with diatoms-cyanobacteria as the sub-dominant groups. The relative contribution of chrysophyte biomass to the phytoplankton assemblage has varied from 2008 to 2013. While diatom biomass exhibited an increasing trend from 2004 to 2009, it has been decreasing in the main basin of Snap Lake. In the northwest arm of Snap Lake, phytoplankton community composition is mainly chrysophyte-dominated, with the relative proportion of diatoms increasing over time. Phytoplankton community composition in Northeast Lake has differed from Snap Lake since sampling began in 2008. Northeast Lake has consistently been a cyanobacteria-chrysophyte dominated lake, while Lake 13 is a chrysophyte-dominated lake.

The changes in the phytoplankton community composition that were observed in the main basin from 2004 to 2009 (i.e., increased overall biomass and increased diatom dominance) appear to now be occurring in the northwest arm of Snap Lake. The lag in response to Mine-related activities is consistent with the slow spread of Mine-related water into the northwest arm of Snap Lake. The main basin is now fully mixed, with no consistent spatial patterns in phytoplankton variables, in relation to proximity to the diffuser. In addition, it has undergone a number of community-level changes since baseline. The decreases in biomass and shift back to a chrysophyte-dominated community may be indicators of the phytoplankton community in the main basin returning to a more stable state, similar to baseline. Mine-related increases in conductivity, N, and Si have taken longer to reach the northwest arm of Snap Lake; therefore, changes to the phytoplankton community in the northwest arm are now being observed.

Toxicity testing demonstrated stimulation in algal growth (i.e., *Pseudokirchneriella subcapitata*), with the degree of stimulation increasing at higher sample concentrations of Snap Lake water (Appendix 3F), rather than inhibition. These toxicity test results are in agreement with what is being observed in the phytoplankton community, i.e., despite a decrease in phytoplankton biomass, a negligible enrichment effect in Snap Lake rather than toxicological impairment affect is apparent.

5.6.3 Key Question 3: What is the Current Status, in Terms of Abundance, Biomass and Composition, of the Zooplankton Community in Snap Lake, Northeast Lake, and Lake 13, and do these Results Suggest Signs of Mine-Related Nutrient Enrichment or Toxicological Impairment?

Mean annual total zooplankton abundance varied over time in Snap Lake and Northeast Lake. In the main basin of Snap Lake total zooplankton abundance has decreased since baseline (i.e., 2004); in 2013, it was 1.6 times lower than the baseline value. Mean annual total zooplankton biomass has decreased over time in Snap Lake; from 2009 to 2013, total zooplankton biomass has remained below baseline values. In the northwest arm of Snap Lake, total zooplankton biomass has decreased over time from 2006 to 2013, with the exception of an increase in 2010. Conversely, zooplankton biomass in Northeast Lake has increased since 2008. Overall, total zooplankton biomass was similar among stations in 2013.

In terms of relative zooplankton biomass, the zooplankton community in Snap Lake and Northeast Lake has been consistently dominated by calanoid copepods. However, in the main basin of Snap Lake, the community has undergone a community shift from a calanoid copepod-dominated community during baseline, to a cyclopoid-calanoid copepod co-dominated community from 2007 to 2012, back to a mainly calanoid copepod-dominated community in 2013. In the northwest arm of Snap Lake, calanoid copepods decreased in dominance from 2004 to 2009, and increased in dominance from 2009 to 2013, following the trend observed in the main basin of Snap Lake. Zooplankton community composition in Northeast Lake was similar to Snap Lake; however, composition in Lake 13 differed, i.e., rotifers-dominated in 2012, but a more even distribution among the major groups was observed in 2013. This difference in 2012 may be a reflection of Lake 13 being sampled only in August.

Toxicity testing results with cladocerans demonstrated that there were no adverse effects for any of the test endpoints, e.g., *Daphnia magna* or *Ceriodaphnia dubia* in Snap Lake in 2013. These tests suggest that the decreases observed in the zooplankton community since baseline may not be caused by toxicological impairment, but rather are natural fluctuations in the zooplankton community.

5.6.4 Key Question 4: How do Observed Changes Compare to Applicable Predictions in the EAR?

The EAR predicted that chlorophyll *a* concentrations would gradually increase from 0.20 to 1.8 µg/L to 1.5 to 2.3 µg/L over a 20-year period, with chlorophyll *a* concentrations remaining within the range associated with oligotrophic lakes and without a change in the overall productive status of Snap Lake. In 2013, the overall mean annual chlorophyll *a* concentration in the main basin and the northwest arm of Snap Lake remained within the range characteristic of oligotrophic lakes (0.30 to 4.5 µg/L; Wetzel 2001).

The phytoplankton community results indicate that changes have occurred since baseline conditions (i.e., 2004). Changes in mean total phytoplankton biomass and abundance between 2004 and 2013 are of relatively low magnitude. The EAR prediction of negligible increases in phytoplankton biomass and abundance is consistent with the low magnitude changes observed in phytoplankton biomass and abundance to date. Changes in community composition are evident at both the major group level and the genus level. Overall, the results are inconsistent with the EAR prediction of a minor change in phytoplankton community structure. The EAR predicted a change in the relative proportion of various species, which has been observed. The EAR also predicted that no loss of species or major shifts in keystone species was expected; the results to date show that minor shifts at the group-level have occurred.

The zooplankton community results indicate that changes have occurred since baseline conditions (i.e., 2004) in Snap Lake. But the changes in mean total zooplankton biomass and abundance between 2004 and 2013 are of relatively low magnitude. The EAR predicted a slight increase in zooplankton abundance and biomass, which could lead to a minor increase in fish food and a minor change in zooplankton community structure. It stated that the relative proportion of various species may change, but no loss of species and no major shifts in keystone species are expected. The EAR prediction of negligible increases is consistent with the low magnitude increases that were initially observed in zooplankton biomass and abundance in Snap Lake to date. The EAR also predicted that no loss of species or major shifts in keystone species was expected; the results to date show that minor shifts at the group-level have occurred.

5.7 Recommendations

Based on the results to date, no changes are required for the plankton program other than a re-evaluation of the Golder plankton QC procedures (Section 5.3.2).

5.8 References

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SECTION 6

BENTHIC INVERTEBRATE COMMUNITY

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LIST OF ACRONYMS

Term	Definition
AEMP	Aquatic Effects Monitoring Program
ANOVA	analysis of variance
DO	dissolved oxygen
EAR	Environmental Assessment Report
GPS	global positioning system
i.e.	that is
Mine	Snap Lake Mine
MVLWB	Mackenzie Valley Land and Water Board
n	number of stations
n/a	not applicable
n/s	not significant
NAD	North America Datum
NMDS	non-metric multidimensional scaling
P-value	probability
PAH	polycyclic aromatic hydrocarbons
SD	standard deviation
SE	standard error
TDS	total dissolved solids
TOC	total organic carbon
UTM	Universal Transverse Mercator

UNITS OF MEASURE

Term	Definition
°C	degrees Celsius
%	percent
±	plus or minus
m	metre
µm	micrometre
µS/cm	microSiemens per centimetre
mg	milligram
mg/L	milligrams per litre
m^2	square metres
no./m^2	number per square metre
organisms/m^2	organisms per square metre
taxa/station	taxa per station
ww	wet weight

6 BENTHIC INVERTEBRATE COMMUNITY

6.1 Introduction

The assessment endpoints for the Aquatic Effects Monitoring Program (AEMP) are based on the valued ecosystem components identified in the Environmental Assessment Report (EAR), the effect predictions in the EAR, and narrative commitments made by De Beers during the EAR process (De Beers 2002) and through the Environmental Agreement (De Beers 2004). De Beers committed that water quality, fish health, and ecological function will remain acceptable in Snap Lake. The benthic invertebrate community monitoring component of the AEMP addresses the ecological function assessment endpoint.

6.1.1 Background

Benthic invertebrates are small aquatic animals that lack backbones; they live on the bottoms of waterbodies such as lakes and streams. Freshwater benthic invertebrates include mostly insect larvae, crustaceans, worms, leeches, snails, and clams. They form diverse communities often consisting of thousands of individuals per square metre (m^2). Benthic invertebrates live on the surface of the sediments or burrow into sediments, although some species are closely associated with aquatic plants and are frequently sampled to monitor the environmental quality of lakes for the following reasons (Rosenberg and Resh 1993):

- they are present in nearly all waterbodies and are usually abundant;
- they remain in a small area throughout the aquatic phase of their life cycle;
- they obtain food by various means including the filtering of fine particulates and feeding on algae, decaying organic material, aquatic plants, or other invertebrates;
- they have relatively long life cycles ranging from months to years, thereby integrating the effects of disturbances over a relatively long period;
- they are an important food source for organisms at higher trophic levels such as fish;
- they are sensitive to a variety of disturbances, including: addition of sediment, toxins, nutrients, and organic material; low dissolved oxygen (DO) levels; and, alteration of flow, substratum, and temperature;
- they respond to disturbances in a predictable manner;
- they can be relatively easily collected and identified; and,
- the wide range of species inhabiting any given location means that animals of varying sensitivity are present.

This section presents benthic invertebrate community data and habitat data collected in September 2013 in Snap Lake, Northeast Lake, and Lake 13. Benthic invertebrate community characteristics were summarized and benthic community variables were compared statistically among these lakes.

The community in Snap Lake, which is exposed to treated effluent from the Snap Lake Mine (Mine), was compared to reference lakes communities in Northeast Lake and Lake 13. In addition, the variation over time in benthic community variables was evaluated using 2009 to 2013 data.

The 2013 benthic invertebrate program represents the ninth year of benthic invertebrate community monitoring under the AEMP, and is based on the 2013 AEMP Design Plan (De Beers 2014). The initial AEMP was submitted in 2005 (De Beers 2005a) and modified in 2013 as required by the Water Licence MV2011L2-0004 (MVLWB 2013). Baseline data were collected in fall 1999 (De Beers 2002) and late winter 2004 (De Beers 2005b). The first year of monitoring occurred in 2005 (De Beers 2006). Input from the Environmental Assessment Report (EAR) (De Beers 2002), permitting hearings, regulators, and the community were used to design the initial AEMP (De Beers 2005a). Changes were made after analysis of the 2005 benthic invertebrate monitoring results indicated that water depth was a confounding factor that interfered with the detection of potential Mine-related effects.

The 2013 AEMP benthic invertebrate community program represents the fifth year of open-water sampling during fall. The benthic invertebrate program was moved from late winter to fall in 2009 due to logistical issues associated with winter field work, which prevented completion of the benthic program in some years. Also, decreased DO in areas of Snap Lake exposed to treated effluent, which was the original reason for sampling under ice, was not observed during winter.

Benthic invertebrate samples were collected in the following lakes, according to the control/impact sampling design described in the 2013 AEMP Design Plan (De Beers 2014):

- Snap Lake:
  - Northwest Arm (three stations);
  - Main Basin (seven stations);
- Reference lakes:
  - Northeast Lake (five stations); and,
  - Lake 13 (five stations).

6.1.2 Objectives

Benthic invertebrate community monitoring is conducted to evaluate the health of the benthic invertebrate community in Snap Lake. The benthic invertebrate community survey is designed to address Water Licence (MVLWB 2013) Schedule 6, Part G (1a, vii), which requires an evaluation of the effects on the benthic invertebrate community due to changes in water or sediment quality in Snap Lake, and Schedule 6, Part G, which requires monitoring the deep water benthic invertebrate community to verify the EAR predictions relating to the trophic and DO status of Snap Lake.

The objective of the 2013 Snap Lake benthic invertebrate community survey was to address the following two key questions:

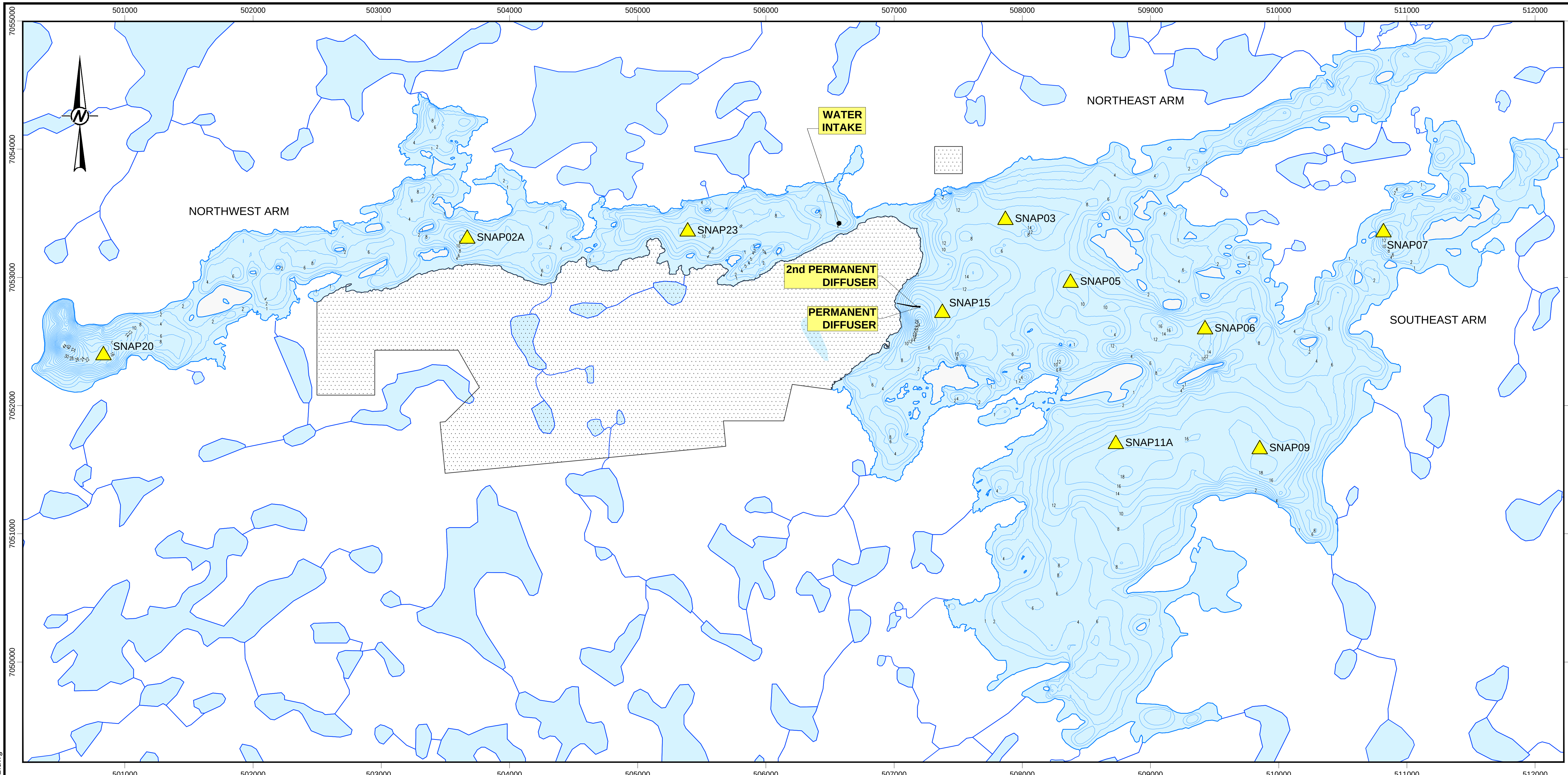
- In 2013, was the benthic invertebrate community affected by the changes in water and sediment quality in Snap Lake?
- If the benthic invertebrate community was affected, was the change greater than stated in the EAR?

6.2 Methods

6.2.1 Field Survey

6.2.1.1 Study Area

The study area includes the main basin and the northwest arm of Snap Lake (Figure 6-1), and two reference lakes referred to as Northeast Lake (Figures 6-2) and Lake 13 (Figure 6-3). Gaps in station numbering in Figures 6-1 to 6-3 occur because a common, comprehensive numbering system was used, which applies to all monitoring programs in Snap Lake. The missing numbers were used by other programs, such as the water quality monitoring program.



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LEGEND

- Depth Contour (m)
- Watercourse
- Waterbody
- Snap Lake Mine Footprint
- Benthic Invertebrate Monitoring Station

REFERENCES

Digital Map from Mackay lake, Northwest Territories, produced by Department of Energy, Mines and Resources. Map 75M, Original scale 1:250,000, projection : Transverse Mercator, Datum : NAD83, coordinate system : UTM zone 12.

Snap Lake outline and islands were corrected to LANDSAT 7 satellite image 45/15, dated September 2, 2000. provided by Geobase.

Bathymetry was created in Surfer 8 using sonar data from the 2002 North Lakes program (Golder) and 2005 transect data from the Reference Lake Search program (Golder).

SNAP LAKE MINE

Benthic Invertebrate Monitoring Stations in Snap Lake, 2013

PROJECTION: UTM Zone 12
DATUM: NAD83

500 0 500
SCALE 1:30,000 METRES

DE BEERS
GROUP OF COMPANIES

FILE No: 1313490001-7010_6-1
DATE: February 24, 2014
JOB No: 14-1349-0004
REVISION No: A
OFFICE: GOLDER VICTORIA
DRAWN: JEF
CHECK: PMC

Figure 6-1

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LEGEND

- Depth Contour (m)
- Watercourse
- Waterbody
- Benthic Invertebrate Monitoring Station

REFERENCES

Digital Map from Mackay lake, Northwest Territories, produced by Department of Energy, Mines and Resources. Map 75M, Original scale 1:250,000, projection : Transverse Mercator, Datum : NAD83, coordinate system : UTM zone 12.

Lake outline and islands were corrected to LANDSAT 7 satellite image 45/15, dated September 2, 2000. provided by Geobase.

Bathymetry was created in Surfer 8 using sonar data from the 2002 North Lakes program (Golder) and 2005 transect data from the Reference Lake Search program (Golder).

NOTES

Grid is displayed in Transverse Mercator, Datum : NAD83,
Coordinate system : UTM zone 12.

SNAP LAKE MINE

Benthic Invertebrate Monitoring Stations in Northeast Lake, 2013

PROJECTION: UTM Zone 12	DATUM: NAD83		
SCALE 500 0 500 METRES		DE BEERS GROUP OF COMPANIES	
FILE No: 1313490001-7010_6-2		DATE: February 24, 2014	
JOB No: 14-1349-0004	REVISION No: A	Figure 6-2	
OFFICE: GOLDER VICTORIA	DRAWN: JEF	CHECK: PMC	

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LEGEND

Depth Contour (m)

Watercourse

Waterbody

Benthic Invertebrate Monitoring Station

REFERENCES

Digital Map from Mackay lake, Northwest Territories, produced by Department of Energy, Mines and Resources. Map 75M, Original scale 1:250,000, projection : Transverse Mercator, Datum : NAD83, coordinate system : UTM zone 12.

Bathymetry was created in Surfer 8 using sonar data from the 2002 North Lakes program (Golder) and 2005 transect data from the Reference Lake Search program (Golder).

SNAP LAKE MINE

Benthic Invertebrate Monitoring Stations in Lake 13, 2013

PROJECTION: UTM Zone 12	DATUM: NAD83	DE BEERS GROUP OF COMPANIES
500 0 500 SCALE METRES		
FILE No: 1313490001-7010_6-3		DATE: February 24, 2014
JOB No: 14-1349-0004	REVISION No: A	Figure 6-3
OFFICE: GOLDER VICTORIA	DRAWN: JEF	CHECK: PMC

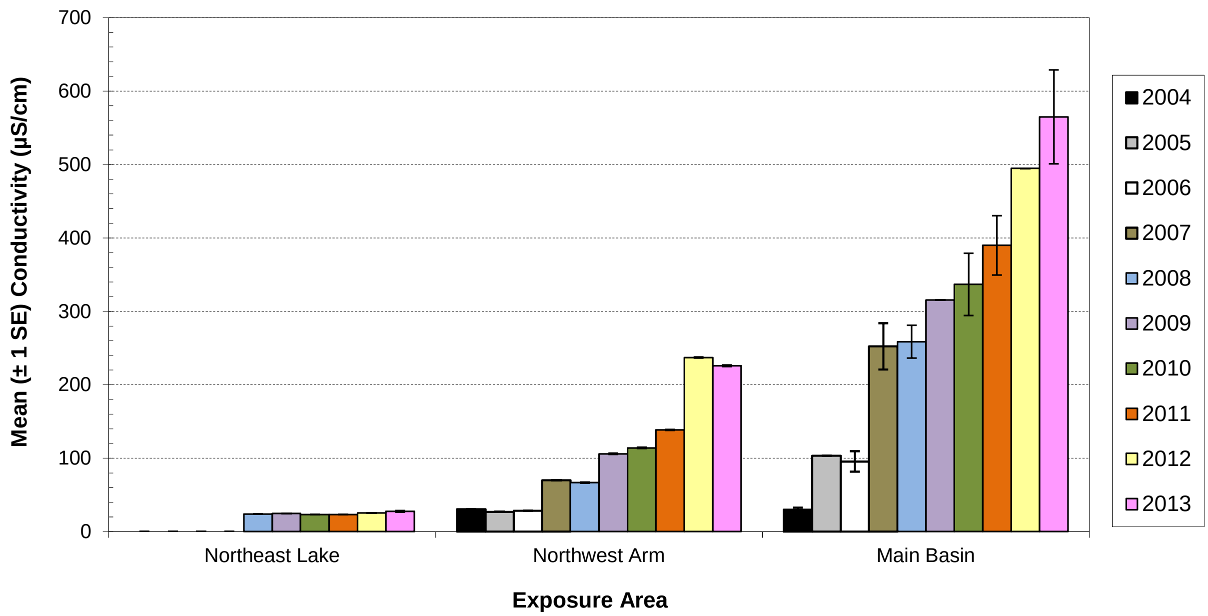
6.2.1.2 Study Design

The benthic invertebrate community survey is conducted every three years, as outlined in the updated 2013 AEMP design plan (De Beers 2014). If necessary, increased frequency of the benthic invertebrate sampling program could be triggered by results of annual water quality and sediment quality monitoring, the level of effects detected during the AEMP benthic study, or substantive changes to Mine operations. The 2013 sampling program was triggered due to a potential decreasing trend in total density, *Microtendipes* density and Pisidiidae density, and a continued decrease in richness in 2012. Near-field, mid-field, and far-field areas from previous AEMP monitoring programs were combined in the updated 2013 AEMP study design (De Beers 2014) into a single exposure area Snap Lake (main basin), because these three areas are similarly exposed to treated effluent as indicated during both late-winter and fall conductivity measurements (Appendix 6A, Figure 6A-2). The number of stations sampled in the main basin has been reduced to seven from the ten originally sampled. In the main basin, benthic invertebrate samples were collected at the same stations as the water quality component, with the following exceptions:

- SNAP15 was sampled in place of water quality stations SNP02-20e because SNP02-20e is deeper than the maximum depth of 15 m required for benthic invertebrate sampling.
- SNAP07 was added to the monitoring program to monitor the benthic invertebrate community near the outlet of Snap Lake. This station was sampled in place of water quality monitoring station SNAP08 because SNAP08 is shallower than the minimum depth of 10 metres (m) required for benthic invertebrate sampling.

Northwest arm stations in Snap Lake continue to be sampled as part of the AEMP, because they are less exposed to treated effluent compared to the main basin (Figure 6-4; Appendix 6A, Figure 6A-2) and are used to monitor the spread of treated effluent in the northwest arm. In the northwest arm, benthic invertebrate stations are the same as those sampled by the water quality component with the exception of SNAP20, which is sampled in place of water quality station SNAP20B, because SNAP20B is deeper than the maximum depth of 15 m selected for benthic invertebrate sampling.

Figure 6-4 Late-Winter Mean Near-Bottom Conductivity in Each Sampling Area in Snap Lake, 2004 to 2013



Note: Bars with no error bars represent data for fewer than three stations.
µS/cm = microSiemens per centimetre; ± = plus or minus; SE = standard error.

Two reference lakes, Northeast Lake and Lake 13, were sampled in 2013 for comparison with the main basin of Snap Lake. Benthic invertebrate samples were collected at five stations in both Northeast Lake and Lake 13.

Benthic invertebrate monitoring stations were located in water depths ranging from 10 to 15 m, as in previous monitoring programs from 2006 to 2012. This depth range was chosen to eliminate depth as a potential confounding factor when analyzing benthic invertebrate data for potential mine-related effects.

6.2.1.3 Sampling Methods

Benthic invertebrate samples were collected during the fall open water program in Snap Lake, Northeast Lake and Lake 13 from September 5 to 15, 2013.

Benthic invertebrates were sampled according to standard operating procedures (Golder 1997). At each station, an Ekman grab was lowered over the side of an anchored boat to obtain benthic samples. Six individual Ekman grabs were collected at each station. Each sample was sieved through a 500 micrometre (µm) mesh Nitex® screen; material retained in the mesh was placed in a separate 1-litre (L) Nalgene® bottle and preserved in 10 percent (%) buffered formalin.

Samples were shipped to J Zloty, Ph.D, located in Summerland, British Columbia, for enumeration and taxonomic identification of invertebrates. For the majority of stations, the six grabs were combined into a single composite sample. At one station within each sampling area (northwest arm and main basin) in Snap Lake, and one station in each reference lake (Northeast Lake and Lake 13), the six grabs were processed as discreet samples. These discreet samples allowed for an estimate of within-station variability, to assess whether six samples per station were sufficient to collect representative samples during the fall survey.

At each station, an additional composite sediment sample, consisting of three grabs, was collected for analyses of total organic carbon (TOC) and particle size. In Lake 13, sediment samples were also analyzed for nutrients, metals, and polycyclic aromatic hydrocarbons (PAHs), as part of the winter road monitoring program. Sediment samples were placed into sample containers provided by the laboratory and shipped in coolers on ice to ALS Canada Ltd. in Edmonton, Alberta, for analyses. Sediment quality results are described in Section 4.

6.2.1.4 Sample Sorting and Taxonomic Identification

Samples were processed according to standard protocols based on recommendations of Environment Canada (2012) and Gibbons et al. (1993). Benthic invertebrate samples were first washed through a 500 micrometre (µm) sieve to remove the preservative and fine sediments remaining after field sieving. Organic material was separated from inorganic material using elutriation, and the inorganic material was checked for any remaining shelled or cased invertebrates, which were removed and added to the organic material. The organic material was split into coarse and fine fractions using a set of nested sieves of 1-millimetre (mm) and 500 µm mesh sizes. As samples were generally small, typically containing less than 100 organisms, laboratory subsampling was not done.

Invertebrates were identified to the lowest practical taxonomic level, typically genus, using recognized taxonomic keys (Soponis 1977; McAlpine et al. 1981; Oliver and Roussel 1983; Wiederholm 1983; Brinkhurst 1986; Pennak 1989; Clifford 1991; Coffman and Ferrington 1996; Wiggins 1996; Kathman and Brinkhurst 1998; Maschwitz and Cook 2000; Epler 2001; Throp and Covich 2001; Merritt et al. 2008). Organisms that could not be identified to the desired taxonomic level, such as immature or damaged specimens, were reported as a separate category at the lowest taxonomic level possible, typically family. Organisms that required detailed microscopic examination for identification, such as midges (Chironomidae) and aquatic worms (Oligochaeta), were mounted on microscope slides using an appropriate mounting medium. Most common taxa were distinguishable based on gross morphology and required only a few slide mounts for verification. All rare or less common taxa were slide mounted for identification.

Invertebrates removed from the samples, sorted organic material, and archived samples are being stored for six years to allow possible comparisons, if necessary, with samples collected during subsequent monitoring.

6.2.1.5 Supporting Environmental Variables

During the benthic invertebrate survey, the following supporting environmental information was recorded at each station:

- sampling date and time;
- weather conditions, such as air temperature, wind velocity, and wind direction;
- global positioning system (GPS) coordinates recorded as Universal Transverse Mercator (UTM);
- water depth; and,
- vertical profiles of water temperature, DO, pH, and conductivity, measured at 1 m intervals.

The UTM co-ordinates were recorded using a hand-held Garmin GPS unit. A YSI 650 Multi-parameter Display System water quality meter with a YSI 600 Quick Sample multi-parameter water quality probe was used to measure field water quality profiles. Additional details of field water quality measurements are provided in Section 3.

6.2.2 Data Analyses

6.2.2.1 Data Entry and Screening

Raw benthic invertebrate abundance data were received from the taxonomist in Microsoft Excel® spreadsheet format, with data entry already verified. Non-benthic organisms, such as cyclopoid copepods (Cyclopoida), and water fleas (Cladocera) were removed from the data before analyses. True fly (Diptera) pupae were also removed before data analyses because the pupal stages of some Dipteran taxa are non-benthic. Abundance data received as number of organisms per sample were converted to density, expressed as the number of organisms per square metre (organisms/m^2). Unusual abundance data were validated before data summary calculations and statistical analyses.

6.2.2.2 Key Question 1: Is the benthic invertebrate community affected by changes in water and sediment quality in Snap Lake?

Benthic Community Variables

The following summary variables were calculated for each station as station means, with the exception of richness, which was expressed as total richness per station:

- total invertebrate density (organisms/m^2);
- community composition as percentages of major taxa;
- taxonomic richness;
- Simpson's diversity index (diversity);

- evenness; and,
- dominance.

Summary statistics including the arithmetic mean, median, minimum, maximum, standard deviation (SD), and standard error (SE) were calculated for each of the above variables.

Eight additional variables were included in statistical comparisons for the 2013 program. Densities of dominant invertebrates based on the 2013 data set, defined as those accounting for more than 5% of total invertebrates across all stations, were compared among sampling areas. These invertebrates were *Valvata sincera* (snails, Gastropoda: Valvatidae), fingernail clams (Pisidiidae), and six Chironomidae genera (*Microtendipes*, *Micropsectra*, *Heterotrissocladius*, *Corynocera*, *Procladius*, and *Tanytarsus*). Together, these taxa accounted for 76% of total invertebrates in the 2013 data set. Biomass was not included in the statistical analyses, because it generally reflects total invertebrate density, but tends to be more variable within and among stations.

Before statistical testing, data were checked for normality using the Shapiro-Wilk test, and for homogeneity of variances using Bartlett's test. Distributions of total density and densities of dominant taxa, except *Microtendipes* density, were found to be significantly non-normal (P <0.05). No significant deviations from normality were detected for other variables (all tests P >0.05). Bartlett's test results identified diversity, *Micropsectra* density, *Valvata sincera* density, *Procladius* density, *Corynocera* density, and *Tanytarsus* density as having significantly heterogeneous variances among sampling areas (P <0.05); all other variables had homogeneous variances in all areas (all tests, P >0.05). Therefore, total density and densities of dominant taxa data, except for *Microtendipes* density, were square root or natural log(x+1) transformed before statistical analyses as appropriate, which eliminated the heterogeneity of variances and the majority of deviations from normality observed in the untransformed data. The exceptions were *Micropsectra* density and *Corynocera* density, where transformations did not eliminate the deviations from normality; these variables were compared among areas using a nonparametric test. In addition, diversity was also compared among areas using a nonparametric test.

Habitat Relationships

Relationships between habitat variables and biological variables were evaluated by calculating Spearman rank correlation coefficients and examining scatter-plots. Correlations were evaluated between the biological variables identified above and the habitat variables water depth, TOC, and the percentage of fine sediments, which consists of the silt and clay particle size fractions. Correlations were run using SYSTAT 13.1 (SYSTAT 2009) and were considered statistically significant at P <0.05.

Among-Area Comparisons

Benthic community variables were compared statistically between the Snap Lake main basin and the two reference lakes. The northwest arm was excluded from these comparisons, because of the varying degree of exposure of stations in this area to treated effluent. The statistical analyses followed environmental effects monitoring data analysis protocols (Environment Canada 2012). The unit of replication was the station. Variables compared statistically were total density, total richness, diversity, evenness, and densities of Pisidiidae, *Valvata sincera*, *Microtendipes, Micropsectra*, *Heterotrissocladius*, *Procladius*, *Corynocera*, and *Tanytarsus*.

One-way analysis of variance (ANOVA) or the Kruskal-Wallis test (Sokal and Rohlf 1995), which is the non-parametric equivalent of ANOVA, were used for an initial overall comparison. Results of these tests were considered significant at P <0.1. After a significant ANOVA or Kruskal-Wallis test result, the following comparisons were conducted using planned orthogonal contrasts (Sokal and Rohlf 1995):

- Northeast Lake and Lake 13 pooled compared to main basin of Snap Lake; and,
- Northeast Lake compared to Lake 13.

Results of contrasts were considered significant at P <0.1.

An additional, unplanned ANOVA or Kruskal-Wallis test comparing only the main basin of Snap Lake to Northeast Lake was conducted for each benthic invertebrate variable. This comparison was conducted due to the differences observed in the benthic invertebrate community in Lake 13 compared to the main basin of Snap Lake and Northeast Lake in 2012 and 2013, which suggested that Lake 13 is not suitable for direct comparisons to the Snap Lake main basin.

Comparison to Normal Range and Evaluation of Trends over Time

The mean values for summary variables for the Snap Lake main basin were plotted with the normal range from Northeast Lake overlaid, to determine whether any of the variables were outside the normal range. Normal ranges were calculated as the mean ±2 SD using Northeast Lake data for each station for 2009 to 2013 based on fall data. Only Northeast Lake data were used to calculate normal ranges, because the addition of Lake 13 data increased the upper limit of the normal range to the point where detecting an enrichment effect would be unlikely. Trends over time in the main basin of Snap Lake were also evaluated visually using these plots.

Statistical Power

For benthic invertebrate monitoring, the recommended critical effect size is ±2 SD, estimated from reference area data (Environment Canada 2012). Using this effect size and a significance level of α=0.1, generic power analysis results provided by Environment Canada (2012) indicate that the five stations per area are sufficient to achieve the desired power of 0.9. A retrospective power analysis was conducted on non-significant among-area ANOVA comparisons to check that this level of power was achieved.

To illustrate the magnitude of the chosen critical effect size, ±2 SDs were calculated based on 2009 to 2013 fall data for Northeast Lake for each summary variable, and were expressed as the percentage of the mean.

Multivariate Analysis

Non-metric multidimensional scaling (NMDS; Kruskal 1964; Cox and Cox 2001) was run on the benthic invertebrate data to summarize community structure and evaluate potential differences in community structure among Snap Lake, Northeast Lake, and Lake 13. Non-metric multidimensional scaling is a nonparametric ordination method that allows for the reduction of a data set consisting of a large number of taxa to typically two dimensions referred to as ordination axes (Clarke 1993). The analysis is based on a station-by-station distance matrix and provides a visual representation of ecological distances among stations.

A station-by-station Bray-Curtis distance matrix was generated from the ln(x+1) transformed density data and was used as the input for the ordination. Two dimensions were selected for the ordination, after confirming that the stress value of the two-dimensional configuration was reasonably low (less than 0.2; Clarke 1993). This analysis was run using SYSTAT 13.1 (SYSTAT 2009). Ordination results were presented as a two-dimensional scatter-plot of the sampling stations in ordination space.

6.2.2.3 Key Question 2: If the benthic invertebrate community is affected, is the change greater than predicted in the EAR?

If changes in the benthic invertebrate community were observed, an evaluation of the statistical and visual results was used to determine whether the change in the benthic community was within EAR predictions. This evaluation was based on the magnitude of change observed and considered whether results from multiple evaluation methods indicated a change.

6.3 Quality Assurance and Quality Control

6.3.1 Benthic Invertebrate Taxonomy

Invertebrate sample sorting efficiency was verified by an individual other than the original sorter by performing spot-checks on sediment remaining after sorting (the debris). Ten percent of the samples were re-sorted. The data quality objective was a minimum removal of 90% of the total number of organisms in a sample. If more than 10% of the total number of organisms removed from the sample were found in the debris, then all samples were re-sorted by an individual other than the original sorter. In addition, if an entire taxonomic group was inadvertently omitted by the sorter, then all samples were re-sorted by an individual other than the original sorter. Removal efficiency was 100% for all samples selected for spot-checks (Appendix 6A, Table 6A-5), indicating that the data quality objective was met.

6.3.2 Data Entry

In accordance with Golder Associates Ltd.'s standard quality assurance and quality control protocol, 10% of all data entered electronically were reviewed for data entry errors. If errors were found in this sub-sample, all data entered electronically were reviewed and corrections were made as appropriate. Supporting data entered from field data sheets were quality checked independently by a second person.

Calculations performed during the data summary and analysis stage were spot-checked for potential errors, and appropriate logic checks were performed to evaluate the accuracy of calculations.

6.4 Results

6.4.1 Supporting Environmental Variables

At the stations sampled for benthic invertebrates during the fall program, water depth ranged from 9 to 15 m (Table 6-1). Water quality parameters at benthic invertebrate stations varied little with depth, indicating that Snap Lake, Northeast Lake, and Lake 13 were well-mixed at these stations during the fall 2013 benthic invertebrate program. Water quality profile data were not available for station SNAP20 for the fall 2013 sampling period.

Fall 2013 conductivity measurements were well above background levels at Snap Lake main basin stations, indicating the presence of treated effluent (Table 6-1, Figure 6-4). At these stations, conductivity was relatively constant throughout the water column (Appendix 6A, Table 6A-4). Conductivity at northwest arm stations SNAP02A and SNAP23 was also above background at 157 and 265 microSiemens per centimetre (µS/cm), respectively, compared to the baseline range of 22 to 36 µS/cm based on 2004 data. Conductivity in Northeast Lake and Lake 13 was similar to background concentrations observed in Snap Lake before 2005, at 22 µS/cm at all stations in Northeast Lake and at 19 to 20 µS/cm in Lake 13.

The fall 2013 conductivity data indicate that treated effluent discharged through the diffuser has reached the entire northwest arm and treated effluent concentration continues to gradually increase in a westerly direction over time.

Inorganic content of bottom sediments consisted of a mixture of silt and clay, with smaller amounts of sand at most stations in Snap Lake. Bottom sediments at Snap Lake stations consisted mostly of fine sediments, with fines content ranging from 96% to 100% (Table 6-2). The composition of bottom sediments in Northeast Lake and Lake 13 was similar, ranging from 96% to 100% fines, and 96% to 99% fines, respectively.

The TOC values for Lake 13 were lower than in both Snap Lake and Northeast Lake in fall 2013. Total organic carbon ranged from 12% to 22% in Snap Lake, from 15% to 18% in Northeast Lake, and from 8% to 11% in Lake 13. The TOC values in Snap Lake and Northeast Lake sediments were relatively high for oligotrophic northern lakes, consistent with low bottom DO measured under background conditions in deep areas of Snap Lake (De Beers 2002).

Due to the low ranges of variation in water depth and sediment particle size distribution, these variables were not expected to interfere with the analysis of Mine-related effects. However, TOC differences among lakes may be large enough to influence the analysis of Mine-related effects; therefore, habitat variation was considered when interpreting results of reference and exposure area comparisons.

Table 6-1 Station Locations and Field Water Quality Parameters Measured in Northeast Lake, Lake 13, and Snap Lake, Fall 2012

Lake	Area	Station	Date	UTM Coordinates		Maximum Depth (m)	Profile Depth (m)	Water Temperature (°C)	Dissolved Oxygen (mg/L)	Specific Conductivity (µS/cm)	pH
				Easting	Northing						
Northeast Lake	-	NEL01	10-Sep-2013	508410	7058967	12	11	10.6	11.0	22	5.5
		NEL02	10-Sep-2013	510098	7058916	11	10	11.0	11.0	22	6.2
		NEL03	10-Sep-2013	510227	7058563	10	10	10.9	10.6	22	6.6
		NEL04	11-Sep-2013	510045	7059742	14	13	10.8	10.9	22	5.9
		NEL05	11-Sep-2013	511479	7059529	9	9	10.8	10.9	22	6.1
					Mean	11	11	10.8	10.9	22	6.1
					Median	11	10	10.8	10.9	22	6.1
					Minimum	9	9	10.6	10.6	22	5.5
					Maximum	14	13	11.0	11.0	22	6.6
Lake 13	-	LK13-01	15-Sep-2013	486983	7063595	12	12	10.3	11.0	20	5.6
		LK13-02	15-Sep-2013	490773	7061874	10	9	10.1	11.3	19	6.6
		LK13-03	15-Sep-2013	492494	7061886	13	12	10.3	11.1	19	6.0
		LK13-04	15-Sep-2013	492964	7061111	11	10	10.4	11.1	19	5.8
		LK13-05	15-Sep-2013	492220	7060984	15	14	10.3	11.0	19	6.4
					Mean	12	11	10.3	11.1	19	6.1
					Median	12	12	10.3	11.1	19	6.0
					Minimum	10	9	10.1	11.0	19	5.6
					Maximum	15	14	10.4	11.3	20	6.6
Snap Lake	Northwest Arm	SNAP02A	05-Sep-2013	503665	7053297	11	10	10.2	10.0	157	7.4
		SNAP20	05-Sep-2013	500830	7052396	15	- [(a)]	- [(a)]	- [(a)]	- [(a)]	- [(a)]
		SNAP23	08-Sep-2013	505381	7053368	11	10	10.9	11.2	265	6.4
	Main Basin	SNAP03	07-Sep-2013	507867	7053461	13	12	11.2	11.1	484	7.2
		SNAP05	07-Sep-2013	508378	7052965	14	13	11.2	11.0	489	7.0
		SNAP06	07-Sep-2013	509424	7052605	13	12	11.3	11.0	480	6.9
		SNAP07	05-Sep-2013	510816	7053354	12	11	10.4	11.3	456	6.2
		SNAP09	07-Sep-2013	509868	7051670	15	14	11.4	10.9	465	6.2
		SNAP11A	07-Sep-2013	508596	7051766	14	13	11.2	10.7	465	6.7
		SNAP15	08-Sep-2013	507363	7052728	12	11	11.2	11.2	506	6.0
					Mean	13	12	11.0	10.9	419	6.7
					Median	13	12	11.2	11.0	465	6.7
					Minimum	11	10	10.2	10.0	157	6.0
					Maximum	15	14	11.4	11.3	506	7.4

Notes: Field water quality data are from near the sediment water interface at the depth indicated in the profile depth column. UTM coordinates are North American Datum (NAD) 83, Zone 12 V.

a) Field water quality profile not collected at SNAP20 in 2013.

- = not applicable or data not available; NEL = Northeast Lake; LK13 = Lake 13; SNAP = Snap Lake; m = metre; °C = degrees Celsius; mg/L = milligrams per litre; % = percent; µS/cm = microSiemens per centimetre; UTM = Universal Transverse Mercator; UTM coordinates are North American Datum (NAD) 83, Zone 12 V.

Table 6-2 Water Depth, Sediment Total Organic Carbon and Inorganic Particle Size in Northeast Lake, Lake 13, and Snap Lake, Fall 2013

Lake	Area	Station	Maximum Depth (m)	Organic Carbon (%)	Sediment Particle Size			
					Sand (%)	Silt (%)	Clay (%)	Fines (Silt + Clay) (%)
Northeast Lake	-	NEL01	12	18	4	78	19	96
		NEL02	11	15	1	75	24	99
		NEL03	10	15	4	76	20	96
		NEL04	14	16	1	79	20	100
		NEL05	9	16	4	76	20	96
	Mean		11	16	3	77	21	98
	Median		11	16	4	76	20	96
	Minimum		9	15	1	75	16	96
	Maximum		14	18	4	79	24	100
Lake 13	-	LK13-01	12	10	1	74	25	99
		LK13-02	10	10	4	77	19	96
		LK13-03	13	8	1	82	17	99
		LK13-04	11	11	1	74	25	99
		LK13-05	15	8	3	75	22	98
	Mean		12	9	2	76	22	98
	Median		12	10	1	75	22	99
	Minimum		10	8	1	74	17	96
	Maximum		15	11	4	82	25	99
Snap Lake	Northwest Arm	SNAP02A	11	21	1	80	20	99
		SNAP20	15	12	1	81	18	99
		SNAP23	11	20	2	76	22	98
	Main Basin	SNAP03	13	19	0	77	22	100
		SNAP05	14	19	3	79	18	97
		SNAP06	13	18	2	80	18	98
		SNAP07	12	22	3	81	16	97
		SNAP09	15	16	1	72	26	99
		SNAP11A	14	17	5	73	22	96
		SNAP15	12	18	1	75	24	99
	Mean		13	18	2	78	20	98
	Median		13	18	2	79	20	98
	Minimum		11	12	0	72	13	96
	Maximum		15	22	5	84	26	100

Note: Sediment particle size data are based on dry weight analysis.
- = not applicable; m = metre; % = percent; NEL = Northeast Lake; LK13 = Lake 13; SNAP = Snap Lake.

6.4.2 Benthic Invertebrate Community Summary Variables

Total Density

Total invertebrate density was variable but generally low at stations sampled in fall 2013, with a whole-lake mean of 771 organisms/m² in Snap Lake, 790 organisms/m² in Northeast Lake, and 3,211 organisms/m² in Lake 13 (Table 6-3, Figure 6-5; raw data are provided in Appendix 6A, Table 6A-1). Total density ranged from 108 to 2,320 organisms/m² in Snap Lake, from 273 to 2,205 organisms/m² in Northeast Lake, and from 1,724 to 4,684 organisms/m² in Lake 13. Maximum densities were observed at SNAP05 in the main basin of Snap Lake, at NEL03 in Northeast Lake, and at LK13-05 in Lake 13. Density was highly variable among stations in all three lakes. Densities in Lake 13 were generally higher compared to both Northeast Lake and Snap Lake.

Table 6-3 Benthic Invertebrate Summary Variables in Northeast Lake, Lake 13, and Snap Lake, Fall 2013

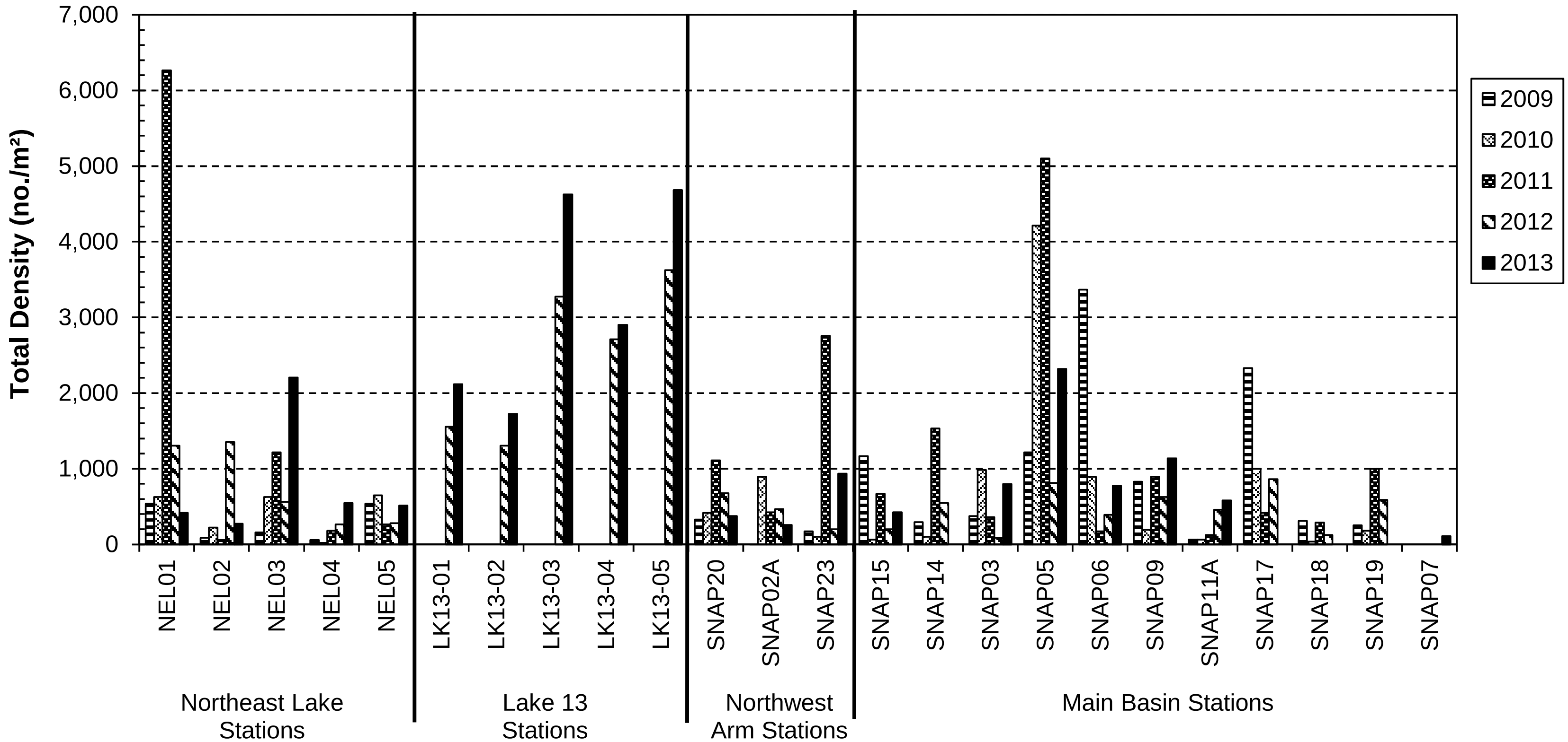
Lake	Area	Station	Total Density (no./m²)	Total Richness (taxa/station)	Simpson's Diversity Index	Evenness
Northeast Lake	-	NEL01	417	9	0.62	0.29
		NEL02	273	12	0.85	0.55
		NEL03	2,205	19	0.78	0.24
		NEL04	546	14	0.62	0.19
		NEL05	510	15	0.78	0.31
	Mean		790	14	0.73	0.32
	Median		510	14	0.78	0.29
	Minimum		273	9	0.62	0.19
	Maximum		2,205	19	0.85	0.55
Lake 13	-	LK13-01	2,119	29	0.90	0.34
		LK13-02	1,724	19	0.89	0.48
		LK13-03	4,626	29	0.85	0.24
		LK13-04	2,902	24	0.90	0.42
		LK13-05	4,684	23	0.83	0.26
	Mean		3,211	25	0.87	0.35
	Median		2,902	24	0.89	0.34
	Minimum		1,724	19	0.83	0.24
	Maximum		4,684	29	0.90	0.48

Table 6-3 Benthic Invertebrate Summary Variables in Northeast Lake, Lake 13, and Snap Lake, Fall 2013

Lake	Area	Station	Total Density (no./m²)	Total Richness (taxa/station)	Simpson's Diversity Index	Evenness
Snap Lake	Northwest Arm	SNAP02A	259	10	0.78	0.45
		SNAP20	374	12	0.74	0.32
		SNAP23	934	12	0.72	0.29
	Main Basin	SNAP03	799	14	0.77	0.31
		SNAP05	2,320	21	0.85	0.31
		SNAP06	776	12	0.82	0.47
		SNAP07	108	5	0.77	0.88
		SNAP09	1,135	15	0.72	0.24
		SNAP11A	582	12	0.79	0.39
		SNAP15	424	9	0.82	0.62
	Mean		771	12	0.78	0.43
	Median		679	12	0.78	0.36
	Minimum		108	5	0.72	0.24
	Maximum		2,320	21	0.85	0.88

- = not available; no./m^2 = number per square metre; taxa/station = taxa per station; NEL = Northeast Lake; LK13 = Lake 13; SNAP = Snap Lake.

Figure 6-5 Total Benthic Invertebrate Density in Northeast Lake, Lake 13, and Snap Lake, Fall 2009 to 2013



Note: Main basin stations are arranged in order along the likely treated effluent flow path based on lake bathymetry.
no./m^2= number per square metre; NEL = Northeast Lake; LK13 = Lake 13; SNAP = Snap Lake.

Community Composition

The dominant benthic taxa in Snap Lake during fall sampling were the Chironomidae, accounting for 20% to 89% of the total density at all stations (Table 6-4, Figure 6-6), with all but four stations having Chironomidae representing greater than 50% of the total density. All four stations with the Chironomidae accounting for less than 50% of the total density are in the main basin of Snap Lake. Pisidiidae were also abundant, accounting for up to 42% of total density. At station SNAP07, the “Other taxa” category accounted for 67% of the total density, and consisted mostly of Nematoda (33%) and Oligochaeta (20%). Station SNAP07 is located in the northeast arm near the outlet of Snap Lake and is closer to the shoreline compared to the other stations in the main basin. The majority of the Chironomidae density consisted of the Chironomini and Tanytarsini tribes. Dominance of the benthic community by the Chironomidae is expected in the sub-Arctic region where Snap Lake is located (Beaty et al. 2006; Northington et al. 2010). The main basin of Snap Lake had a higher proportion of Pisidiidae compared to Northeast Lake and Lake 13 in 2013.

Richness

Richness values in Snap Lake, Northeast Lake, and Lake 13 in 2013 were within the expected range for lake habitats in the sub-Arctic region, with occasional low values in Snap Lake and Northeast Lake. Richness was similar in Northeast Lake and Snap Lake, but higher in Lake 13, ranging from 5 to 21 taxa/station in Snap Lake, from 9 to 19 taxa/station in Northeast Lake, and from 19 to 29 taxa/station in Lake 13 (Table 6-3, Figure 6-7). Richness was significantly positively correlated with total density ($r = 0.882$; $P<0.001$). Overall, the fall 2013 richness values for all lakes were similar to those during previous years, and were generally higher than 2012 values for Northeast Lake and Snap Lake.

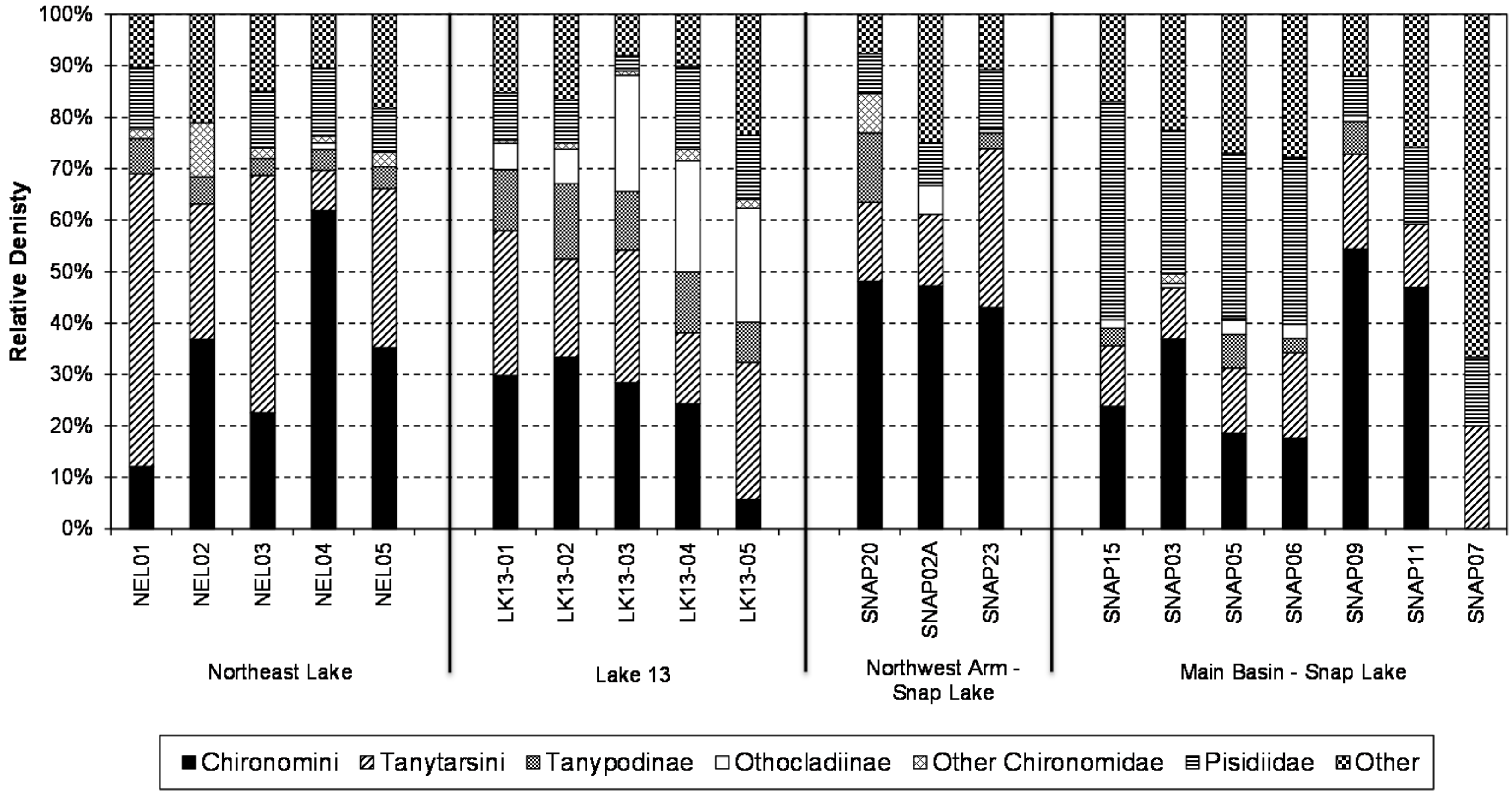
Table 6-4 Relative Densities of Dominant Taxa in Northeast Lake, Lake 13, and Snap Lake, Fall 2013

Taxon	Northeast Lake					Lake 13				
	NEL01 (%)	NEL02 (%)	NEL03 (%)	NEL04 (%)	NEL05 (%)	LK13-01 (%)	LK13-02 (%)	LK13-03 (%)	LK13-04 (%)	LK13-05 (%)
Pisidiidae	12.1	0.0	11.1	13.2	8.5	9.2	8.3	3.0	16.1	12.4
Tanypodinae	6.9	5.3	3.3	3.9	4.2	11.9	14.6	11.3	11.9	7.8
Chironomini	12.1	36.8	22.5	61.8	35.2	29.8	33.3	28.4	24.3	5.7
Tanytarsini	56.9	26.3	46.3	7.9	31.0	28.1	19.2	25.8	13.9	26.7
Orthocladiinae	0.0	0.0	0.0	1.3	0.0	5.1	6.7	22.7	21.5	22.1
Other Chironomidae	1.7	10.5	2.0	1.3	2.8	0.7	1.3	0.8	2.2	1.8
Other	10.3	21.1	15.0	10.5	18.3	15.3	16.7	8.1	10.1	23.5
Total	**100**	**100**	**100**	**100**	**100**	**100**	**100**	**100**	**100**	**100**
Total Chironomidae	77.6	78.9	73.9	76.3	73.2	75.6	75.0	89.0	73.8	64.1

Taxon	Northwest Arm - Snap Lake			Main Basin – Snap Lake						
	SNAP02A (%)	SNAP20 (%)	SNAP23 (%)	SNAP03 (%)	SNAP05 (%)	SNAP06 (%)	SNAP07 (%)	SNAP09 (%)	SNAP11 (%)	SNAP15 (%)
Pisidiidae	8.3	7.7	11.5	27.9	32.2	32.4	13.3	7.6	14.8	42.4
Tanypodinae	0.0	13.5	3.1	0.0	6.5	2.8	0.0	6.3	0.0	3.4
Chironomini	47.2	48.1	43.1	36.9	18.6	17.6	0.0	54.4	46.9	23.7
Tanytarsini	13.9	15.4	30.8	9.9	12.7	16.7	20.0	18.4	12.3	11.9
Orthocladiinae	5.6	0.0	0.8	0.9	2.8	2.8	0.0	1.3	0.0	1.7
Other Chironomidae	0.0	7.7	0.0	1.8	0.3	0.0	0.0	0.0	0.0	0.0
Other	25.0	7.7	10.8	22.5	26.9	27.8	66.7	12.0	25.9	16.9
Total	**100**	**100**	**100**	**100**	**100**	**100**	**100**	**100**	**100**	**100**
Total Chironomidae	66.7	84.6	77.7	49.5	40.9	39.8	20.0	80.4	59.3	40.7

% = percent; NEL = Northeast Lake; LK13 = Lake 13; SNAP = Snap Lake.

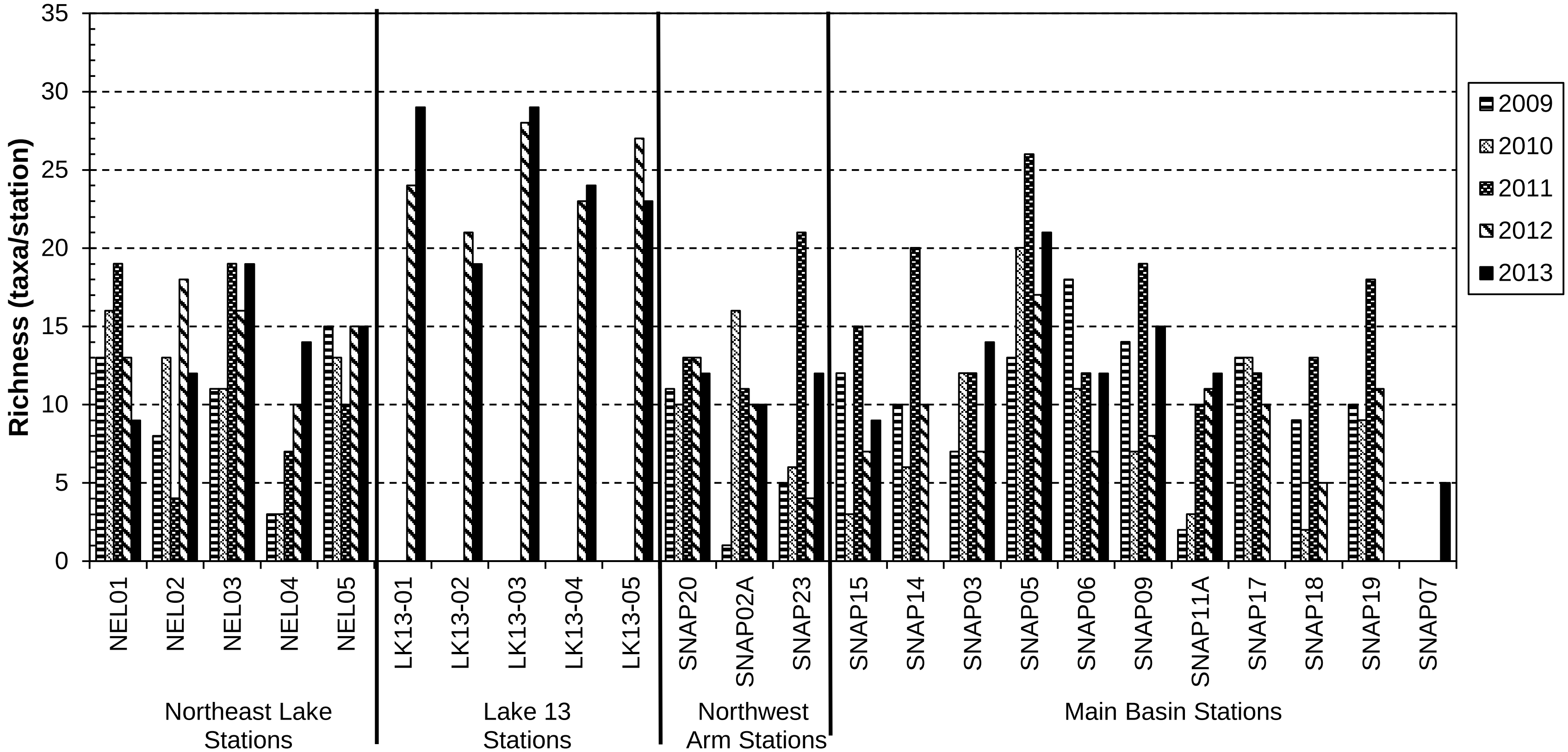
Figure 6-6 Benthic Invertebrate Community Composition in Northeast Lake, Lake 13, and Snap Lake, Fall 2013



Note: Main basin stations are arranged in order along the likely treated effluent flow path based on lake bathymetry.

% = percent; NEL = Northeast Lake; LK13 = Lake 13; SNAP = Snap Lake.

Figure 6-7 Taxonomic Richness of the Benthic Invertebrate Community in Northeast Lake, Lake 13, and Snap Lake, Fall 2009 to 2013



Note: Main basin stations are arranged in order along the likely treated effluent flow path based on lake bathymetry.

NEL = Northeast Lake; LK13 = Lake 13; SNAP = Snap Lake.

Simpson's Diversity Index

Diversity values varied from 0.72 to 0.85 in Snap Lake in fall 2013 (Table 6-3, Figure 6-8), implying a high level of diversity. Diversity values in Northeast Lake ranged from 0.62 to 0.85, and in Lake 13 ranged from 0.83 to 0.90 in fall 2013, indicating a similar level of diversity to Snap Lake. There were no obvious Mine-related differences in diversity between Northeast Lake, Lake 13, and Snap Lake.

Evenness

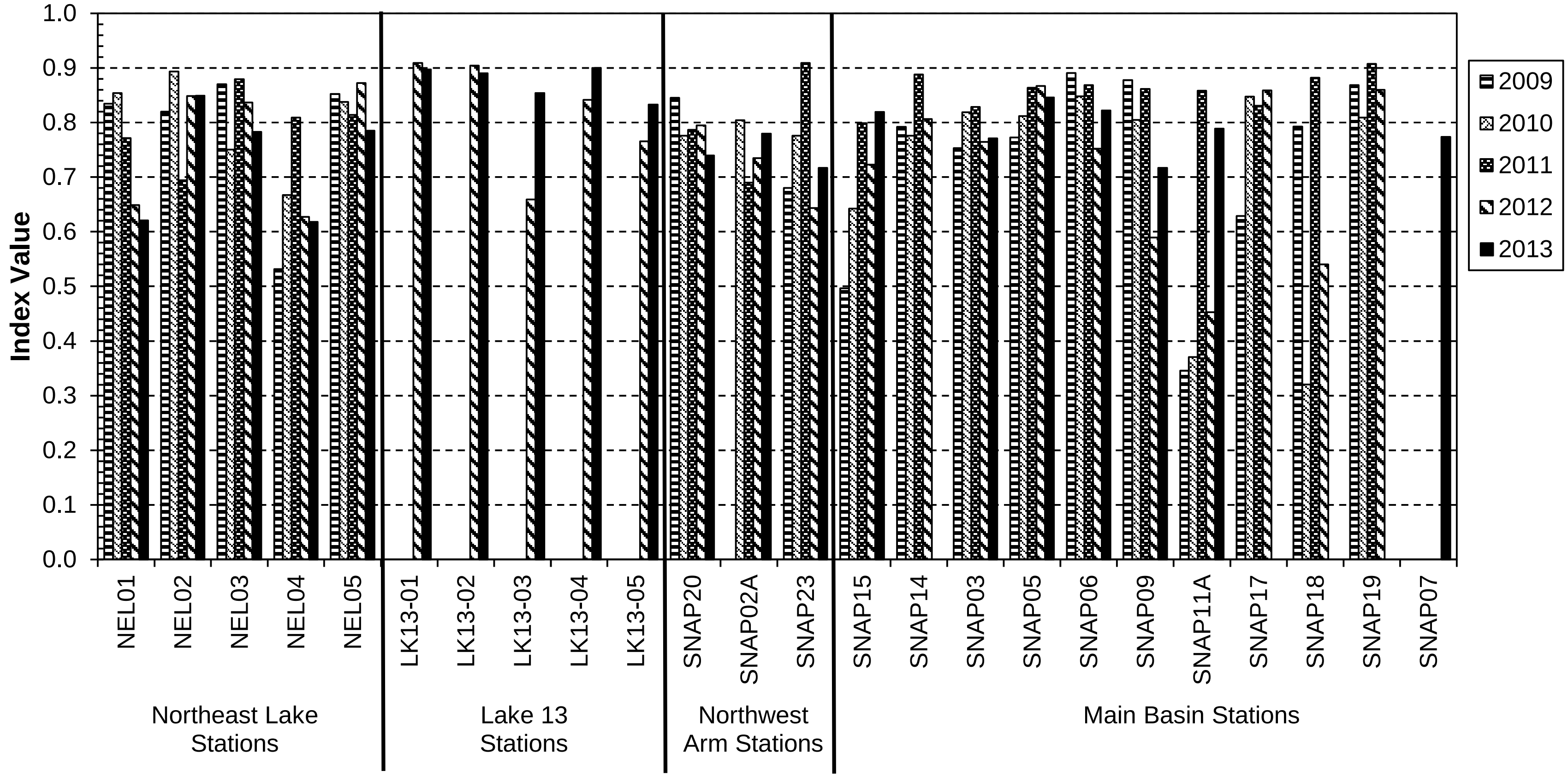
Evenness varied from 0.24 to 0.88, with a mean of 0.43 in Snap Lake in fall 2013 (Table 6-3, Figure 6-9). Snap Lake stations had generally low to moderate evenness. In Snap Lake, high evenness values (greater than 0.60) were generally observed at stations with low total density, and low richness. Evenness was similar in Northeast Lake, ranging from 0.19 to 0.55, with a mean of 0.32, and in Lake 13, ranging from 0.24 to 0.48, with a mean of 0.35 in fall 2013. There were no obvious Mine-related differences in evenness between Northeast Lake and any of the sampling areas in Snap Lake.

Biomass

Invertebrate biomass was low in Snap Lake and Northeast Lake, as expected for the habitat and geographic area sampled. Biomass was higher in Lake 13 compared to both Snap Lake and Northeast Lake in 2013. Total benthic invertebrate biomass was highly variable among stations in Snap Lake in fall 2013, ranging from 48 to 1,063 milligrams (mg) per station as wet weight (Table 6-5, Figure 6-10; see Appendix 6A, Table 6A-2 for raw data). This represents an approximately 22-fold range in invertebrate biomass among stations. The highest biomass of 1,063 mg was observed at SNAP05 in the main basin of Snap Lake and the second highest biomass of 384.5 mg was observed at SNAP03 also in the main basin of Snap Lake. Biomass in Northeast Lake was also variable among stations, ranging from 58.2 to 320 mg per station as wet weight. In Lake 13, biomass was less variable among stations compared to Snap Lake and Northeast Lake, ranging from 439 to 653 mg per stations as wet weight. Mollusca (Gastropoda/Pelecypoda) were the major contributor to total biomass, likely due to shell weight, followed by the Chironomidae. In Northeast Lake, the Amphipoda were also a major contributor to total biomass in 2013.

Mean benthic invertebrate biomass was significantly correlated with mean density ($r = 0.943$; $P<0.001$). As a result, the spatial pattern in biomass (Figure 6-10) mirrored that of total density (Figure 6-6). The likely reason for this strong correlation is the dominance of Pisidiidae and Chironomidae in the benthic community in Snap Lake, combined with the low degree of size variation within these families.

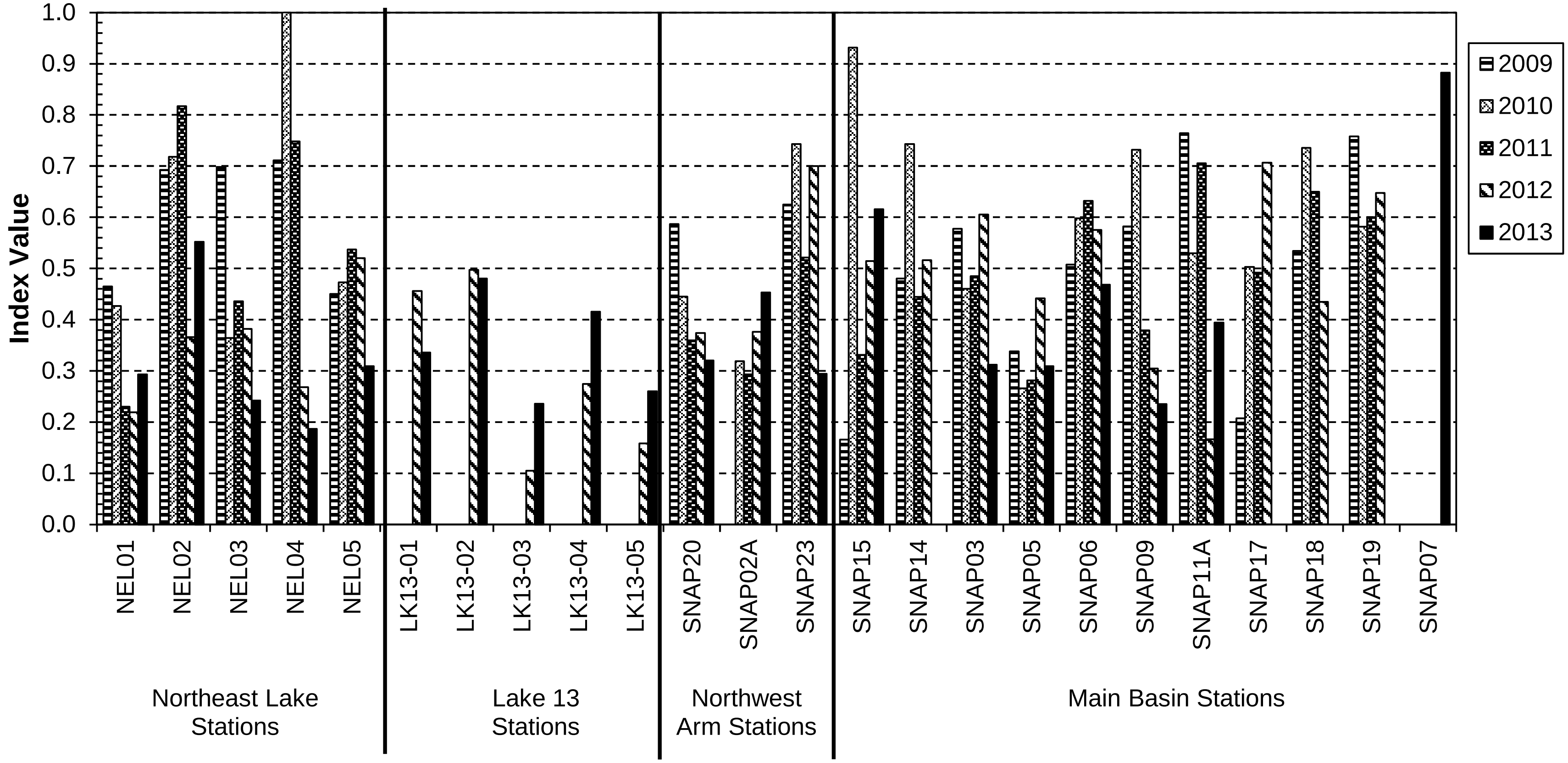
Figure 6-8 **Simpson's Diversity Index Values for the Benthic Invertebrate Community of Northeast Lake, Lake 13, and Snap Lake, Fall 2009 to 2013**



Notes: Main basin stations are arranged in order along the likely treated effluent flow path based on lake bathymetry. No value is provided for SNAP02A in 2009 because only one taxon was collected and calculating diversity based on a single taxon is not valid.

NEL = Northeast Lake; LK13 = Lake 13; SNAP = Snap Lake.

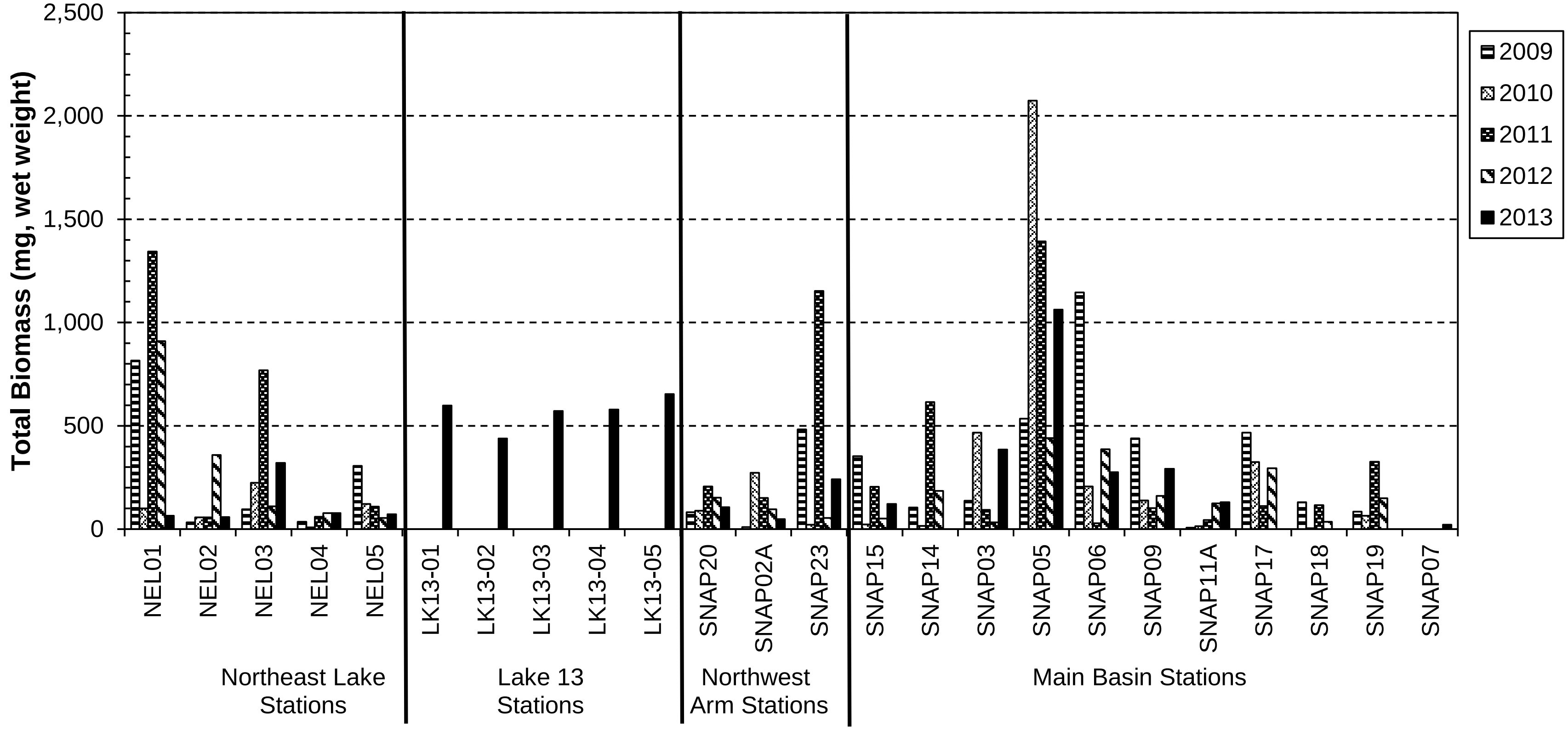
Figure 6-9 Evenness of the Benthic Invertebrate Community of Northeast Lake, Lake 13, and Snap Lake, Fall 2009 to 2013



Notes: Main basin stations are arranged in order along likely treated effluent flow path based on lake bathymetry. No value is provided for SNAP02A in 2009 because only one taxon was collected and calculating evenness based on a single taxon is not valid. Station NEL04 in 2010 had three taxa present with the same density for each taxon which results in an evenness value of 1.0 because the total density is evenly distributed among the existing taxa.

NEL = Northeast Lake; LK13 = Lake 13; SNAP = Snap Lake.

Figure 6-10 Benthic Invertebrate Biomass in Northeast Lake, Lake 13, and Snap Lake, Fall 2009 to 2013



Notes: Near-field stations are arranged in order along the likely treated effluent flow path based on lake bathymetry. Biomass was not analyzed for Lake 13 stations in 2012.
mg = milligram; NEL = Northeast Lake; SNAP = Snap Lake.

Table 6-5 Benthic Invertebrate Biomass in Northeast Lake, Lake 13, and Snap Lake, Fall 2013

Taxon	Lake 13					Northeast Lake				
	LK13-01 [mg, ww]	LK13-02 [mg, ww]	LK13-03 [mg, ww]	LK13-04 [mg, ww]	LK13-05 [mg, ww]	NEL01 [mg, ww]	NEL02 [mg, ww]	NEL03 [mg, ww]	NEL04 [mg, ww]	NEL05 [mg, ww]
Gastropoda/Pelecypoda	328.0	237.0	189.3	326.4	434.7	27.7	0.9	84.2	36.0	13.7
Oligochaeta	32.6	61.5	61.8	8.8	39.9	10.1	4.4	18.5	2.6	3.8
Amphipoda	11.2	0.0	0.0	0.0	0.0	7.7	8.6	87.6	0.0	12.5
Trichoptera	0.0	0.0	1.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Chironomidae	223.6	140.3	309.4	231.8	170.4	18.7	44.1	129.4	37.1	39.3
Other Diptera	1.0	0.0	0.0	0.6	0.0	0.0	0.0	0.0	0.0	0.0
Other taxa	1.3	0.4	8.9	10.6	7.6	0.5	0.2	0.8	1.5	2.1
Total	**597.7**	**439.2**	**571.2**	**578.2**	**652.6**	**64.7**	**58.2**	**320.5**	**77.2**	**71.4**

Taxon	Northwest Arm – Snap Lake			Main Basin – Snap Lake						
	SNAP02A [mg, ww]	SNAP20 [mg, ww]	SNAP23 [mg, ww]	SNAP03 [mg, ww]	SNAP05 [mg, ww]	SNAP06 [mg, ww]	SNAP07 [mg, ww]	SNAP09 [mg, ww]	SNAP11 [mg, ww]	SNAP15 [mg, ww]
Gastropoda/Pelecypoda	19.1	8.2	135.1	305.5	893.7	231.6	19.1	186.0	52.9	92.1
Oligochaeta	0.1	2.8	5.4	10.6	10.0	2.0	1.4	13.7	42.0	5.0
Amphipoda	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Trichoptera	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Chironomidae	28.2	94.6	100.6	67.5	150.6	38.0	0.2	92.2	33.4	24.8
Other Diptera	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Other taxa	0.6	0.2	0.6	1.2	8.7	4.1	0.5	0.8	1.7	0.0
Total	**48.0**	**105.8**	**241.7**	**384.8**	**1,063.0**	**275.7**	**21.2**	**292.7**	**130.0**	**121.9**

mg = milligram; ww = wet weight; % = percent; NEL = Northeast Lake; LK13 = Lake 13; SNAP = Snap Lake.

6.4.3 Correlations with Habitat Variables

Using the entire data set of 20 stations from Snap Lake, Northeast Lake, and Lake 13, total density, richness, diversity, *Micropsectra* density, *Heterotrissocladius* density, and *Procladius* density were significantly negatively correlated with TOC (Table 6-6). *Corynocera* density was significantly positively correlated with TOC. Relationships between benthic invertebrate community variables and TOC were driven by Lake 13 stations having higher densities and lower TOC compared to Snap Lake and Northeast Lake, with no clear relationship present within each lake (Appendix 6A, Figure 6A-2). As a result, TOC was not included as a covariate in the among-area comparisons.

Table 6-6 Spearman Rank Correlations between Benthic Community Variables and Habitat Variables in Northeast Lake, Lake 13, and Snap Lake, Fall 2013

Variable	Water Depth	Total Organic Carbon	Percent Fines (silt + clay)
Correlations Among Habitat Variables			
Water Depth	1	-	-
Total Organic Carbon	-0.053	1	-
Percent Fines (silt + clay)	0.197	0.043	1
Correlations Between Habitat Variables and Benthic Community Variables			
Total Density	0.195	**-0.549**	-0.102
Total Richness	0.104	**-0.720**	-0.020
Simpson's diversity index	-0.218	**-0.579**	-0.075
Evenness	-0.385	0.215	0.035
Microtendipes density	0.133	-0.047	0.107
Pisidiidae density	0.219	-0.240	-0.020
Micropsectra density	-0.139	**-0.831**	-0.134
Heterotrissocladius density	-0.006	**-0.754**	0.016
Valvata sincera density	0.140	-0.209	-0.004
Procladius density	0.153	**-0.729**	-0.105
Corynocera density	0.032	**0.582**	-0.274
Tanytarsus density	0.233	-0.242	0.122

Note: Significant correlations (P <0.05) are shown in **bold** (n = 20; r_s = 0.447).
- = not applicable.

6.4.4 Comparison of Snap Lake to Reference Lakes

Mean benthic invertebrate summary variables for the main basin of Snap Lake and Northeast Lake differed for evenness, Pisidiidae density, *Micropsectra* density, *Valvata sincera* density, *Corynocera* density, and *Tanytarsus* density (Table 6-7, Figures 6-11 and 6-12). In Lake 13, mean benthic invertebrate summary variables were higher than those for the main basin of Snap Lake and Northeast Lake, with the exception of evenness, which was lower compared to the main basin of Snap Lake and similar to Northeast Lake, and *Corynocera* density, which were lower in Lake 13 compared to both Snap Lake and Northeast Lake. Total density and densities of individual taxa were highly variable among stations in each lake, and among lakes (Figures 6-11 and 6-12). High variation in the main basin of Snap Lake resulted from the high densities at SNAP05 and SNAP09. High variation in Northeast Lake resulted from high density at NEL03. High variation in Lake 13 resulted from high densities at LK13-03 and LK13-05.

Table 6-7 Descriptive Statistics for Benthic Community Variables in Snap Lake, Northeast Lake, and Lake 13, Fall 2013

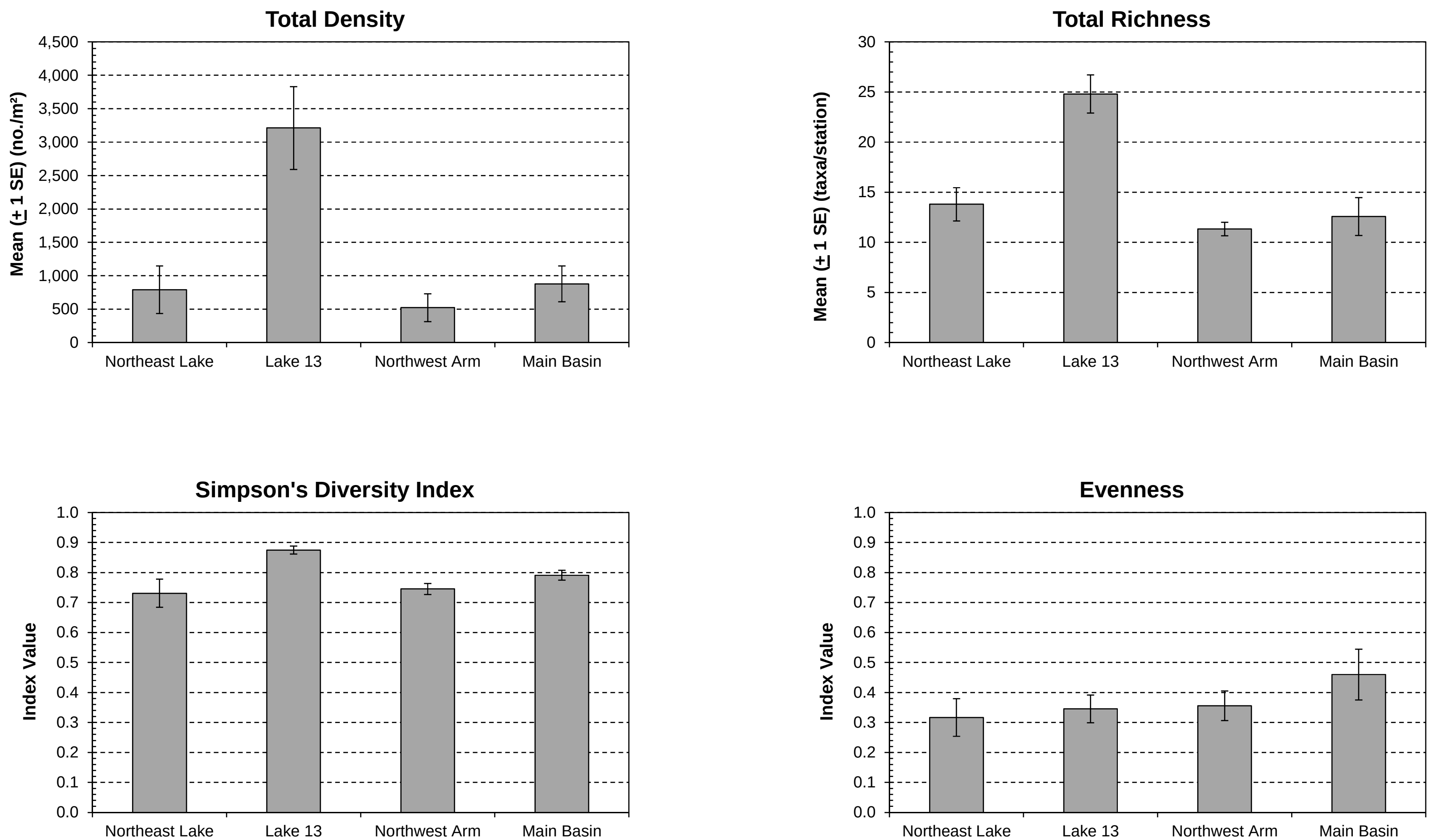
Area	n	Mean	SE	SD	Median	Minimum	Maximum
Total Density (no./m²)							
Northeast Lake	5	790	357	798	510	273	2,205
Lake 13	5	3,211	619	1,385	2,902	1,724	4,684
Northwest Arm	3	522	209	361	374	259	934
Main Basin	7	878	270	713	776	108	2,320
Total Richness (taxa/station)							
Northeast Lake	5	14	2	4	14	9	19
Lake 13	5	25	2	4	24	19	29
Northwest Arm	3	11	1	1	12	10	12
Main Basin	7	13	2	5	12	5	21
Simpson's Diversity Index							
Northeast Lake	5	0.73	0.05	0.11	0.78	0.62	0.85
Lake 13	5	0.87	0.01	0.03	0.89	0.83	0.90
Northwest Arm	3	0.75	0.02	0.03	0.74	0.72	0.78
Main Basin	7	0.79	0.02	0.04	0.79	0.72	0.85
Evenness							
Northeast Lake	5	0.32	0.06	0.14	0.29	0.19	0.55
Lake 13	5	0.35	0.05	0.10	0.34	0.24	0.48
Northwest Arm	3	0.36	0.05	0.09	0.32	0.29	0.45
Main Basin	7	0.46	0.08	0.22	0.39	0.24	0.88
***Microtendipes* Density (no./m²)**							
Northeast Lake	5	197	69	153	151	50	402
Lake 13	5	264	130	291	108	29	740
Northwest Arm	3	170	116	201	108	7	395
Main Basin	7	237	74	196	237	0	568
Pisidiidae Density (no./m²)							
Northeast Lake	5	82	42	94	50	0	244
Lake 13	5	305	92	206	194	136	582
Northwest Arm	3	53	28	48	29	22	108
Main Basin	7	227	92	244	180	14	747
***Micropsectra* Density (no./m²)**							
Northeast Lake	5	82	35	79	57	0	172
Lake 13	5	580	246	549	302	65	1,214
Northwest Arm	3	10	10	17	0	0	29
Main Basin	7	9	9	24	0	0	65

Table 6-7 Descriptive Statistics for Benthic Community Variables in Snap Lake, Northeast Lake, and Lake 13, Fall 2013

Area	n	Mean	SE	SD	Median	Minimum	Maximum
***Heterotrissocladius* Density (no./m²)**							
Northeast Lake	5	0	0	0	0	0	0
Lake 13	5	476	181	404	489	50	876
Northwest Arm	3	0	0	0	0	0	0
Main Basin	7	0	0	0	0	0	0
***Valvata sincera* Density (no./m²)**							
Northeast Lake	5	20	9	21	14	7	57
Lake 13	5	218	108	241	136	72	647
Northwest Arm	3	24	13	22	29	0	43
Main Basin	7	130	61	161	50	14	467
***Procladius* Density (no./m²)**							
Northeast Lake	5	30	9	20	22	14	65
Lake 13	5	322	44	98	330	223	474
Northwest Arm	3	26	15	25	29	0	50
Main Basin	7	37	22	56	14	0	151
***Corynocera* Density (no./m²)**							
Northeast Lake	5	213	156	350	7	0	812
Lake 13	5	1	1	3	0	0	7
Northwest Arm	3	89	78	135	22	0	244
Main Basin	7	23	21	15	29	0	43
***Tanytarsus* Density (no./m²)**							
Northeast Lake	5	10	7	15	7	0	36
Lake 13	5	152	76	170	101	36	445
Northwest Arm	3	24	6	11	22	14	36
Main Basin	7	81	6	66	65	0	187

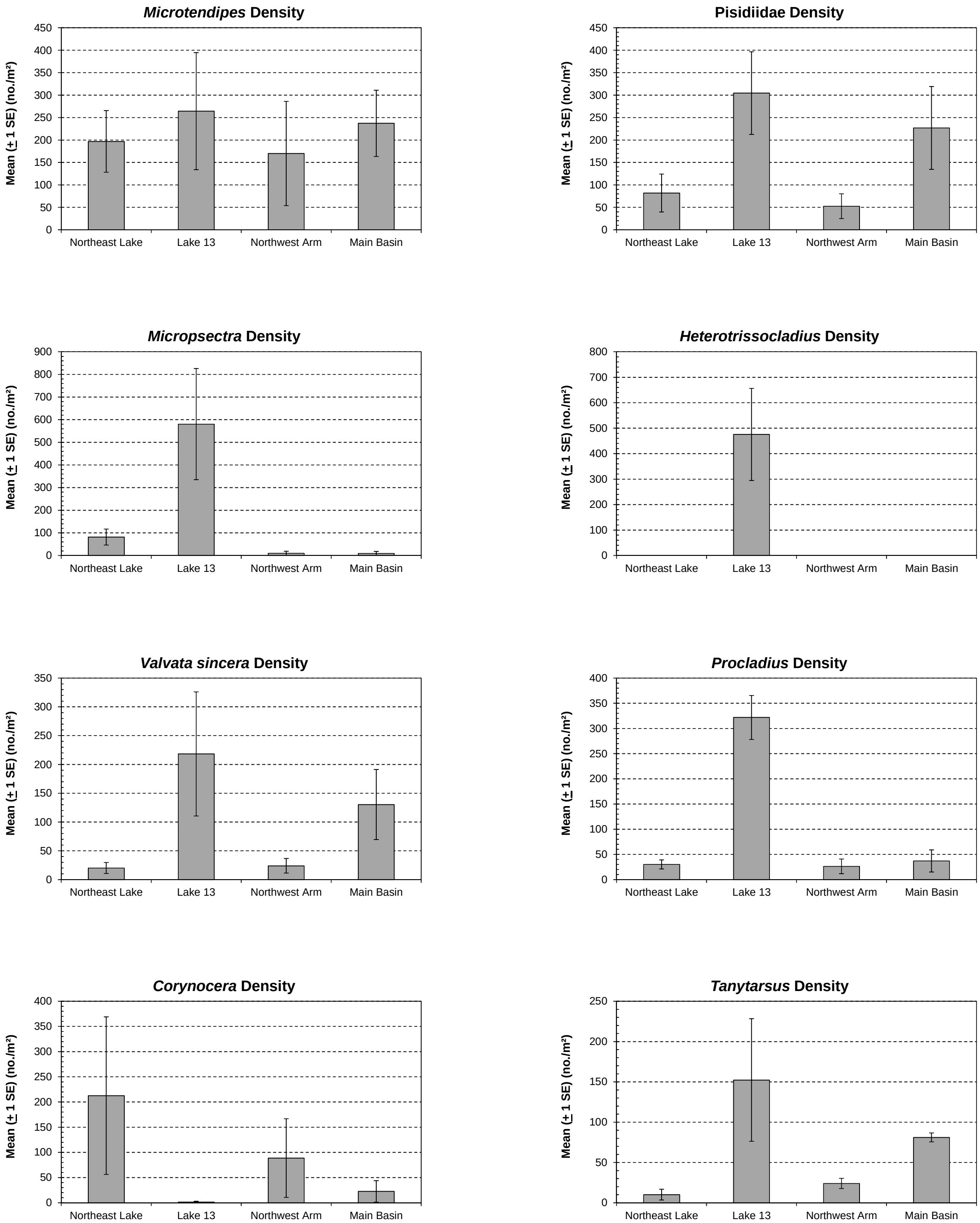
n = number of stations; SE = standard error; SD = standard deviation; no./m2 = number per square metre.

Figure 6-11 Summary Plots for Benthic Community Summary Variables in Snap Lake, Northeast Lake, and Lake 13, Fall 2013



± = plus or minus; SE = standard error; no./m² = number per square metre; taxa/station = taxa per station.

Figure 6-12 Summary Plots for Densities of Dominant Benthic Invertebrates in Snap Lake, Northeast Lake, and Lake 13, Fall 2013



± = plus or minus; SE = standard error; no./m² = number per square metre.

Differences between Snap Lake and Northeast Lake in taxa present were small, with richness ranging from 21 to 27 taxa among areas; however, Lake 13 had a higher richness compared to both lakes, with 40 taxa present (Table 6-8). Only minor differences were apparent among lakes in taxa present within the major taxonomic groups. Oligochaete worms of the sub-family Naidinae were absent from Northeast Lake, but present in both Snap Lake and Lake 13. Amphipoda were present in Northeast Lake and Lake 13, but absent from Snap Lake, and biting midges (Ceratopogonidae*)* were only present in Lake 13. All four major midge groups were represented in the three lakes, with Lake 13 having more Chironomidae genera than Northeast Lake and the main basin of Snap Lake. Also, Lake 13 had a higher number of Orthocladiinae taxa compared to both Snap Lake and Northeast Lake.

Statistical tests comparing benthic community variables among Northeast Lake, Lake 13, and the main basin of Snap Lake detected significant differences in total density, total richness, diversity, Pisidiidae density, *Micropsectra* density, *Valvata sincera* density, *Procladius* density, *Corynocera* density, and *Tanytarsus* density (Table 6-9). Planned comparisons determined that total density, total richness, *Micropsectra* density, and *Procladius* density were significantly lower in the main basin of Snap Lake compared to the pooled reference lakes (Northeast Lake and Lake 13 stations combined). However, statistically significant differences were also detected between Northeast Lake and Lake 13. Total density, total richness, diversity, Pisidiidae density (after the removal of an outlier), *Valvata sincera* density, *Procladius* density, and *Tanytarsus* density were significantly lower in Northeast Lake compared to Lake 13.

Due to the notable differences in the benthic community between the two reference lakes, and between Lake 13 and Snap Lake, Lake 13 is not considered a suitable lake for direct comparison to the main basin of Snap Lake. Northeast Lake is similar to Snap Lake and has been a suitable reference lake for the main basin of Snap Lake over the course of the AEMP. Based on this comparison, *Micropsectra* density was significantly lower in the main basin of Snap Lake compared to Northeast Lake (Table 6-9). *Valvata sincera* density and *Tanytarsus* density were significantly higher in the main basin of Snap Lake compared to Northeast Lake.

The magnitudes of the statistically non-significant comparisons between pooled reference lakes and the main basin of Snap Lake were low for evenness and *Microtendipes* density (<50%). Magnitudes of differences between Northeast Lake and the main basin of Snap Lake were also low for non-significant comparisons, except for Pisidiidae density, *Micropsectra* density, and *Corynocera* density (>50%) suggesting that the sensitivity of statistical tests comparing these variables among sampling areas was low.

The 2013 results confirmed that Lake 13 is not suitable as a reference lake for direct comparison to Snap Lake. As a result, the normal range for the AEMP is based on Northeast Lake data from fall 2009 to fall 2013, as in previous years. This is a conservative approach, because the addition of Lake 13 data would increase the normal range to the point where the upper limit of the range is so high that a slight to moderate enrichment effect would not be detected in the main basin of Snap Lake.

Table 6-8 Presence or Absence of Each Benthic Invertebrate Taxon in Snap Lake, Northeast Lake and Lake 13, Fall 2013

Major Taxon	Family	Subfamily/Tribe	Genus/Species	Northeast Lake	Lake 13	Snap Lake Northwest Arm	Snap Lake Main Basin	Snap Lake Total	All Lakes
Microturbellaria	-	-	-		X		X	X	X
Nematoda	-	-	-	X	X	X	X	X	X
Oligochaeta	Enchytraeidae	-	-				X	X	X
	Lumbriculidae	-	*Lumbriculus*	X	X				X
	Naididae	Naidinae	-		X		X	X	X
		Tubificinae	-	X	X	X	X	X	X
Hydracarina	-	-	-	X	X		X	X	X
Ostracoda	-	-	-	X	X	X	X	X	X
Amphipoda	Hyalellidae	-	*Hyalella azteca*	X	X				X
Gastropoda	Lymnaeidae	-	*Lymnaea*		X		X	X	X
	Valvatidae	-	*Valvata sincera*	X	X	X	X	X	X
Pelecypoda	Pisidiidae	-	(i/d) [(a)]	X	X	X	X	X	X
		-	*Pisidium*	X	X	X	X	X	X
		-	*Sphaerium*	X	X	X	X	X	X
Trichoptera	Phrygaenidae	-	Agrypnia		X				X
Diptera	Ceratopogonidae	Ceratopogoninae	*Bezzia*		X				X
			Probezzia		X				X
	Chironomidae	Tanypodinae	*Ablabesmyia*	X	X				X
			Procladius	X	X	X	X	X	X
			Thienemannimyia gr.		X				X
		Chironomini	*Chironomus*		X		X	X	X
			Cladopelma	X	X	X		X	X
			Cryptochironomus	X	X	X	X	X	X
			Dicrotendipes		X		X	X	X
			Microtendipes	X	X	X	X	X	X
			Pagastiella		X		X	X	X
			Parachironomus		X	X	X	X	X
			Polypedilum		X		X	X	X
			Sergentia		X	X		X	X
			Stictochironomus	X	X	X	X	X	X
		Tanytarsini	*Cladotanytarsus*	X	X	X	X	X	X
			Corynocera	X	X	X	X	X	X
			Micropsectra	X	X	X	X	X	X
			Paratanytarsus			X	X	X	X
			Tanytarsus	X	X	X	X	X	X
		Orthocladiinae	*Cricotopus / Orthocladius*		X				X
			Heterotanytarsus		X				X
			Heterotrissocladius		X				X
			Psectrocladius		X	X	X	X	X
			Zalutschia	X	X				X
		Diamesinae	*Potthastia longimana* gr.	X	X				X
		Diamesinae	*Protanypus*	X	X	X	X	X	X
		Prodiamesinae	*Monodiamesa*	X	X	X		X	X
			Total Taxa	**23**	**40**	**21**	**27**	**30**	**42**

a) Immature and damaged organisms were not included in the total taxa count.

X = present; - = not applicable; i/d = immature or damaged specimen identified to the lowest level possible.

Table 6-9 Results of Statistical Tests Comparing Sampling Areas, Fall 2013

Variable	Test Type	Overall Test Result [(a)] (P-value)	Planned Comparisons [(b)] NEL and LK13 vs Main Basin (P-value)	Planned Comparisons [(b)] NEL vs LK13 (P-value)	ANOVA / K-W NEL vs Main Basin [(e)] (P-value)	Magnitude of Difference NEL and LK13 vs Main Basin (%)	Magnitude of Difference NEL vs LK13 (%)	Magnitude of Difference Main Basin vs NEL (%)	Critical Effect Size NEL and LK13 (%)	Critical Effect Size NEL (%)	Normal Range (NEL 2009 to 2013)
Total density (no./m²)	ANOVA	**0.0077**	**0.0829**	**0.0058**	0.8742	-56	121	11	162	202	0 - 1,572
Total richness (taxa/station)	ANOVA	**0.0008**	**0.0084**	**0.0016**	0.6526	-35	57	-9	79	54	3 - 21
Simpson's diversity index [(c)]	Kruskal-Wallis	**0.0113**	0.2416	**<0.1**	0.4649	-1	18	8	11	29	0.57 - 0.98
Evenness	ANOVA	0.3422	0.1560	0.7963	0.2394	39	9	45	98	88	0.07 - 0.92
Microtendipes density (no./m²)	ANOVA	0.8859	0.9524	0.6313	0.7100	3	29	20	165	156	0 - 316
Pisidiidae density (no./m²)	ANOVA	0.113 / **0.0500**	0.6892 / 0.7147	**0.0431** / **0.0171**	0.1679	17	115	177	215	231	0 - 210
Micropsectra density (no./m²) [(c)]	Kruskal-Wallis	**0.0036**	**0.0039**	n/s	**0.0373**	-97	151	-89	529	193	- [(d)]
Heterotrissocladius density (no./m²)	None	n/a	n/a	n/a	n/a	-100	200	n/a	n/a	n/a	- [(d)]
Valvata sincera density (no./m²)	ANOVA	**0.0089**	0.4760	**0.0029**	**0.0363**	9	166	548	247	211	0 - 130
Procladius density (no./m²)	ANOVA	**<0.0001**	**0.0009**	**0.0001**	0.6277	-79	166	22	305	132	0 - 90
Corynocera density (no./m²) [(c)]	Kruskal-Wallis	**0.0308**	0.1201	n/s	0.8696	-79	197	-89	130	329	- [(d)]
Tanytarsus density (no./m²)	ANOVA	**0.0288** / **0.0330**	0.5706 / 0.1851	**0.0099** / **0.0248**	**0.0294**	0	175	706	162	296	- [(d)]

Note: P-values representing statistically significant differences are **bolded**.

a) ANOVA was used for overall testing unless otherwise indicated. Overall comparisons were considered significant at P <0.1.

b) Planned comparisons for ANOVA tests were considered significant at P <0.1.

c) Tested using Kruskal-Wallis test instead of ANOVA because data transformations did not meet the assumptions for ANOVA. Tests were considered significant at P <0.1, including planned among area comparisons.

d) Taxa were not present in some years from 2009 to 2012.

e) ANOVA or Kruskal-Wallis test comparing Northeast Lake and the main basin of Snap Lake only.

NEL = Northeast Lake; LK13 = Lake 13; Main Basin = main basin of Snap Lake; P-value = probability; % = percent; no./m² = number per square metre; n/s = not statistically significant; n/a = not applicable; ANOVA = analysis of variance; K/W = Kruskal-Wallis.

6.4.5 Statistical Power and Sensitivity

Based on generic power analysis (Environment Canada 2012), a sample size of five stations per area is sufficient to detect an effect size of 2 SD with a power of 0.9 at a significance level of α=0.1, and is appropriate for aquatic effects monitoring. Comparisons of the main basin of Snap Lake, Northeast Lake, and Lake 13 were based on seven stations, five stations, and five stations, respectively. Retrospective power analysis conducted for non-significant ANOVAs comparing benthic community variables among the main basin of Snap Lake, Northeast Lake, and Lake 13 had a power of 0.96 for evenness and 1.00 for *Microtendipes* density for five stations per area. This is a conservative estimate, because there were seven stations sampled in the main basin of Snap Lake, compared to five stations in both Northeast Lake and Lake 13. Results of the power analysis indicate that the power for evenness and *Microtendipes* density were greater than the intended level of 0.90.

In addition to power analysis, sensitivity of statistical tests can also be evaluated qualitatively by comparing the magnitudes of differences among sampling areas to the critical effect size of 2 SD based on reference area data. Statistically significant differences in ANOVA results should detect differences greater than the critical effect size.

To allow an estimate of the critical effect size, which is the limit of background variation, baseline, and reference area data collected 2004 to 2006 for Snap Lake, 2008 to 2013 for Northeast Lake, and 2012 to 2013 for Lake 13, were summarized for the four benthic community summary variables (Table 6-10). The value of 2 SD expressed as the percentage of the reference area mean based on 2004 to 2006 data for Snap Lake, 2008 to 2013 data for Northeast Lake, and 2012 to 2013 data for Lake 13 was largest for total density, ranging from 70% to 332%, and smallest for diversity, ranging from 7% to 36%. The 2 SDs ranged from 24% to 117% for richness, and from 44% to 113% for evenness, which were considered intermediate.

Based on the fall 2013 data, 2 SD as a percentage of the reference area mean was within the range of previous results in both late winter and fall for all benthic invertebrate summary variables (Table 6-10), with the exception of diversity. Although a difference in sampling season may be reflected in the estimate of the critical effect size, the 2 SD values compiled in Table 6-10 using late-winter data are in agreement with those estimated using fall data for at least one other oligotrophic lake (Lac de Gras; DDMI 2009). Unusually high critical effect size estimates were obtained during fall 2009 and 2011 for richness. The unusually high critical effect sizes observed in 2009 for diversity re-appeared in 2012 and have persisted in Northeast Lake in 2013.

Comparisons of the magnitudes of among-area differences in Table 6-9 with the above values revealed that differences between the main basin of Snap Lake and the pooled reference lakes (Northeast Lake and Lake 13) or Northeast Lake were lower than the estimated upper limit of background variation for total density, richness, diversity, and evenness. This comparison indicates that statistical tests were of the desired sensitivity.

Table 6-10 Summary of Baseline, Reference Area, Northeast Lake and Lake 13 Data for Benthic Community Summary Variables, Late Winter 2004 to 2008 and Fall 2009 to 2013

Variable	Year	Area	Season	n	Mean	Median	Range	SD	2 SD (%)[a]
Total Density (no./m^2)	2004	Reference - Snap Lake	Late Winter	8	948	943	362 - 1,377	332	70
	2005	Reference - Snap Lake	Late Winter	5	742	800	373 - 1,290	381	103
	2006	Reference - Snap Lake	Late Winter	6	762	670	186 - 2,014	668	175
	2007	Reference - Snap Lake	Late Winter	2	366	-	237 - 496	-	-
	2008	Northeast Lake	Late Winter	5	796	323	194 - 2,529	982	247
	2009	Northeast Lake	Fall	5	276	158	58 - 540	243	176
	2010	Northeast Lake	Fall	5	429	626	22 - 647	289	135
	2011	Northeast Lake	Fall	5	1,597	266	58 - 6,266	2,650	332
	2012	Northeast Lake	Fall	5	753	561	266 - 1,353	538	143
		Lake 13	Fall	5	2,494	2,712	1,302 - 3,626	1,030	83
	2013	Northeast Lake	Fall	5	790	510	273 - 2,205	798	202
		Lake 13	Fall	5	3,211	2,902	1,724 - 4,684	1,385	86
Richness (no. of taxa)	2004	Reference - Snap Lake	Late Winter	8	13	13	9 - 16	3	43
	2005	Reference - Snap Lake	Late Winter	5	13	12	10 - 17	3	51
	2006	Reference - Snap Lake	Late Winter	6	12	12	10 - 18	3	49
	2007	Reference - Snap Lake	Late Winter	2	10	-	8 - 12	-	-
	2008	Northeast Lake	Late Winter	5	13	13	10 - 16	3	46
	2009	Northeast Lake	Fall	5	10	11	3 - 15	5	94
	2010	Northeast Lake	Fall	5	11	13	3 - 16	4	73
	2011	Northeast Lake	Fall	5	12	10	4 - 19	7	117
	2012	Northeast Lake	Fall	5	14	15	10 - 18	3	43
		Lake 13	Fall	5	25	24	21 - 28	3	24
	2013	Northeast Lake	Fall	5	14	14	9 - 19	4	53
		Lake 13	Fall	5	25	24	19 - 29	4	34
Simpson's diversity index	2004	Reference - Snap Lake	Late Winter	8	0.82	0.85	0.66 - 0.89	0.07	18
	2005	Reference - Snap Lake	Late Winter	5	0.82	0.81	0.73 - 0.90	0.06	15
	2006	Reference - Snap Lake	Late Winter	6	0.82	0.86	0.63 - 0.88	0.10	23
	2007	Reference - Snap Lake	Late Winter	2	0.72	-	0.57 - 0.87	-	-
	2008	Northeast Lake	Late Winter	5	0.86	0.86	0.81 - 0.91	0.04	9
	2009	Northeast Lake	Fall	5	0.78	0.83	0.53 - 0.87	0.14	36
	2010	Northeast Lake	Fall	5	0.80	0.84	0.67 - 0.89	0.09	23
	2011	Northeast Lake	Fall	5	0.79	0.81	0.69 - 0.88	0.07	18
	2012	Northeast Lake	Fall	5	0.77	0.84	0.63 - 0.87	0.12	31
		Lake 13	Fall	5	0.82	0.84	0.66 - 0.91	0.11	27
	2013	Northeast Lake	Fall	5	0.73	0.78	0.62 - 0.85	0.11	29
		Lake 13	Fall	5	0.87	0.89	0.83 - 0.90	0.03	7

Table 6-10 Summary of Baseline, Reference Area, Northeast Lake and Lake 13 Data for Benthic Community Summary Variables, Late Winter 2004 to 2008 and Fall 2009 to 2013

Variable	Year	Area	Season	n	Mean	Median	Range	SD	2 SD (%)[a]
Evenness	2004	Reference - Snap Lake	Late Winter	8	0.50	0.48	0.33 - 0.70	0.12	49
	2005	Reference - Snap Lake	Late Winter	5	0.48	0.38	0.31 - 0.83	0.21	88
	2006	Reference - Snap Lake	Late Winter	6	0.55	0.56	0.25 - 0.77	0.18	67
	2007	Reference - Snap Lake	Late Winter	2	0.46	-	0.29 - 0.63	-	-
	2008	Northeast Lake	Late Winter	5	0.63	0.63	0.33 - 0.90	0.21	67
	2009	Northeast Lake	Fall	5	0.60	0.69	0.45 - 0.71	0.13	44
	2010	Northeast Lake	Fall	5	0.60	0.80	0.32 - 0.85	0.26	87
	2011	Northeast Lake	Fall	5	0.55	0.54	0.23 - 0.82	0.24	87
	2012	Northeast Lake	Fall	5	0.35	0.37	0.22 - 0.52	0.12	69
		Lake 13	Fall	5	0.30	0.27	0.10 - 0.50	0.17	113
	2013	Northeast Lake	Fall	5	0.32	0.29	0.19 - 0.55	0.14	88
		Lake 13	Fall	5	0.35	0.34	0.24 - 0.48	0.10	60

Notes: 2004: baseline data; 2005 to 2007: reference area data (northwest arm); 2008 to 2012 (NEL): reference lake data; 2012 (LK13): preliminary second reference lake data.
Baseline or reference area data include stations with conductivity less than 50 µS/cm and water depth ranging from 8.0 to 16.2 m.
Northeast Lake was included from 2008 onward because the northwest arm of Snap Lake was no longer suitable as a reference area due to exposure to treated effluent.
- = not applicable, median and standard deviation were not calculated because only two stations were sampled in the northwest arm in 2007.
a) 2 SD expressed as the percentage of the baseline, reference area or Northeast Lake mean.
n = number of stations; NEL = Northeast Lake; LK13 = Lake 13; SD = standard deviation; % = percent; µS/cm = microSiemens per centimetre; m = metre; m² = square metres; no./m^2 = number per square metre.

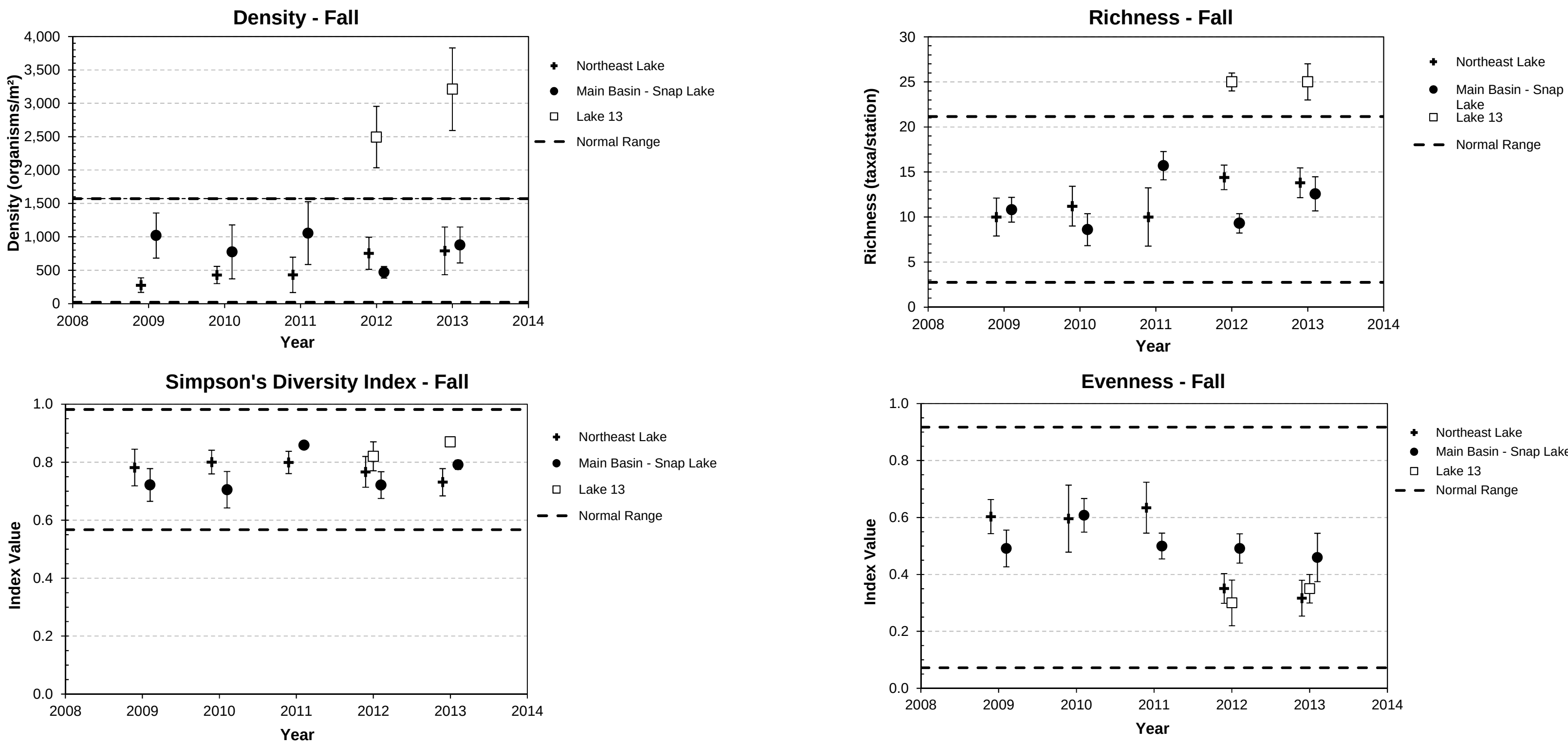
6.4.6 Trends over Time

Trends over time among lakes for benthic invertebrate variables are summarized below, and in Figures 6-13 and 6-14:

- Total density had a slight increasing trend 2009 to 2013 in Northeast Lake and no apparent trend in the main basin of Snap Lake. In Lake 13 there was an increasing trend from 2012 to 2013. Overall, total density increased from 2012 to 2013 in all lakes.
- Total richness had no apparent trend from 2009 to 2012 in both Northeast Lake and the main basin of Snap Lake. In Lake 13 richness was similar between 2012 and 2013.
- No trend was present for Simpson's diversity index in either Northeast Lake or the main basin of Snap Lake from 2009 to 2013, or in Lake 13 from 2012 to 2013.
- No trend in evenness was observed from 2009 to 2011 in Northeast Lake, but a decrease in evenness occurred in 2012 and evenness remained low in 2013. In the main basin of Snap Lake, no trend in evenness was observed from 2009 to 2013. Evenness in Lake 13 was similar between 2012 and 2013.
- *Microtendipes* density had an increasing trend in Northeast Lake and the main basin of Snap Lake from 2011 to 2013. The decreasing trend for *Microtendipes* density in Lake 13 from 2012 to 2013 is likely not real due to the high variability observed in 2012.
- No trend in Pisidiidae density was observed in Northeast Lake compared to a decreasing trend in Pisidiidae density in the main basin of Snap Lake from 2009 to 2012, followed by an increase in 2013. The increase from 2012 to 2013 has brought Pisidiidae density back outside the normal range for the main basin of Snap Lake. Pisidiidae density in the main basin of Snap Lake was also outside the normal range from 2009 to 2011. Pisidiidae density decreased in Lake 13 from 2012 to 2013.
- *Valvata sincera* density increased in the main basin of Snap Lake compared to a slight decrease in Northeast Lake, which may be the beginning of a diverging trend between the main basin of Snap Lake and Northeast Lake.
- *Procladius* density had an increasing trend in Northeast Lake from 2009 to 2012 compared to no trend in the main basin of Snap Lake over the same time period. From 2012 to 2013 *Procladius* density has increase in Lake 13 and decreased in Northeast Lake, while the trend for the main basin of Snap Lake is between the trends for Lake 13 and Northeast Lake.

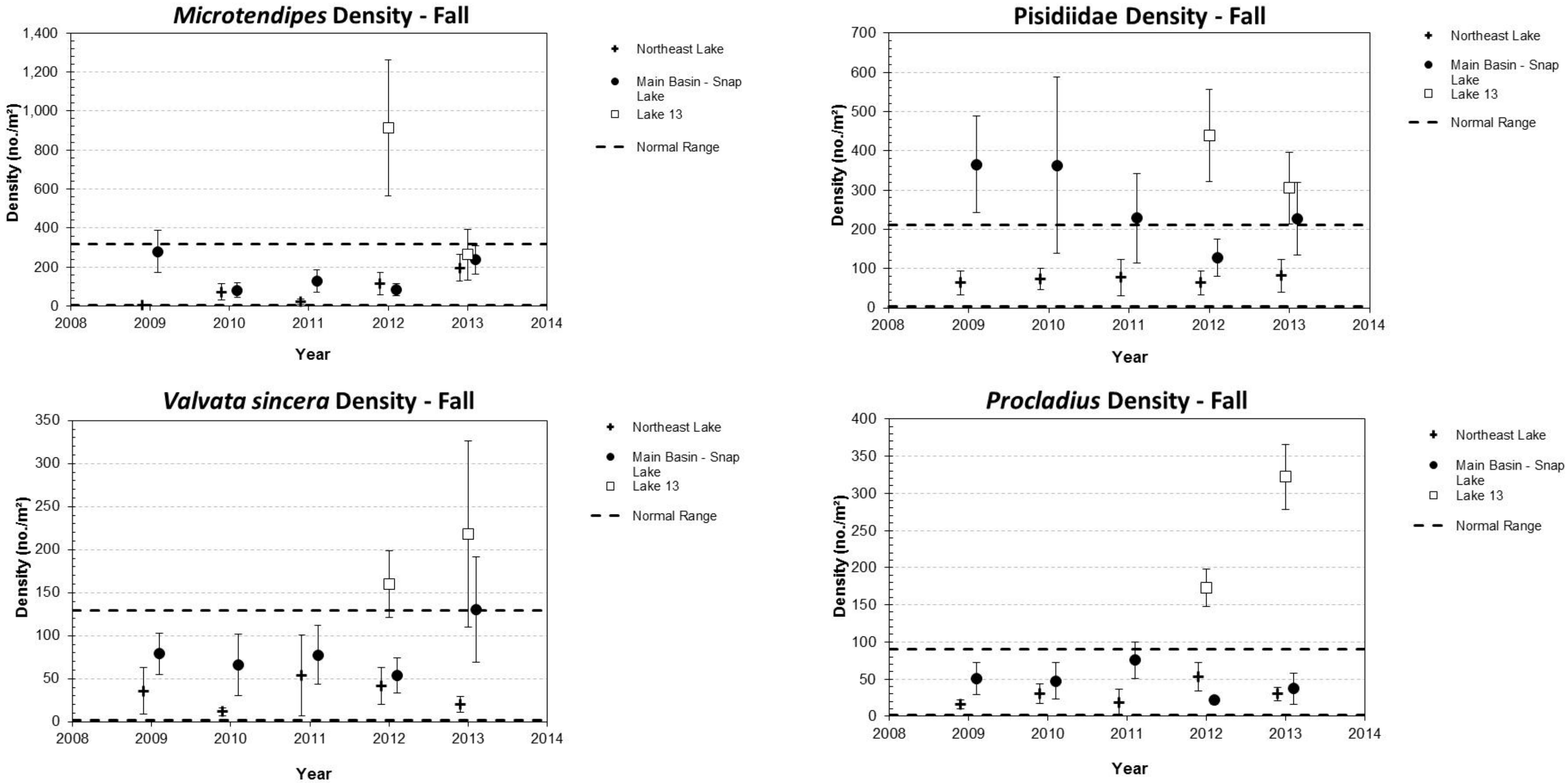
All benthic invertebrate summary variables listed above were within the normal range in Northeast Lake and the main basin of Snap Lake for 2013 (Figures 6-13 and 6-14), with the exception of Pisidiidae density. Pisidiidae density increased from 2012 to 2013, and was outside the normal range in 2013.

Figure 6-13 Annual Means for Benthic Invertebrate Summary Variables in Northeast Lake, Lake 13, and the Main Basin of Snap Lake, 2009 to 2013



Note: Normal range represents mean ± 2 standard deviations based on Northeast Lake station means from 2009 to 2013. Error bars represent ± 1 standard error of the mean.
organisms/m^2 = organisms per square metre; taxa/station = taxa per station.

Figure 6-14 Annual Means for Densities of Common Taxa in Northeast Lake, Lake 13, and the Main Basin of Snap Lake, 2009 to 2013

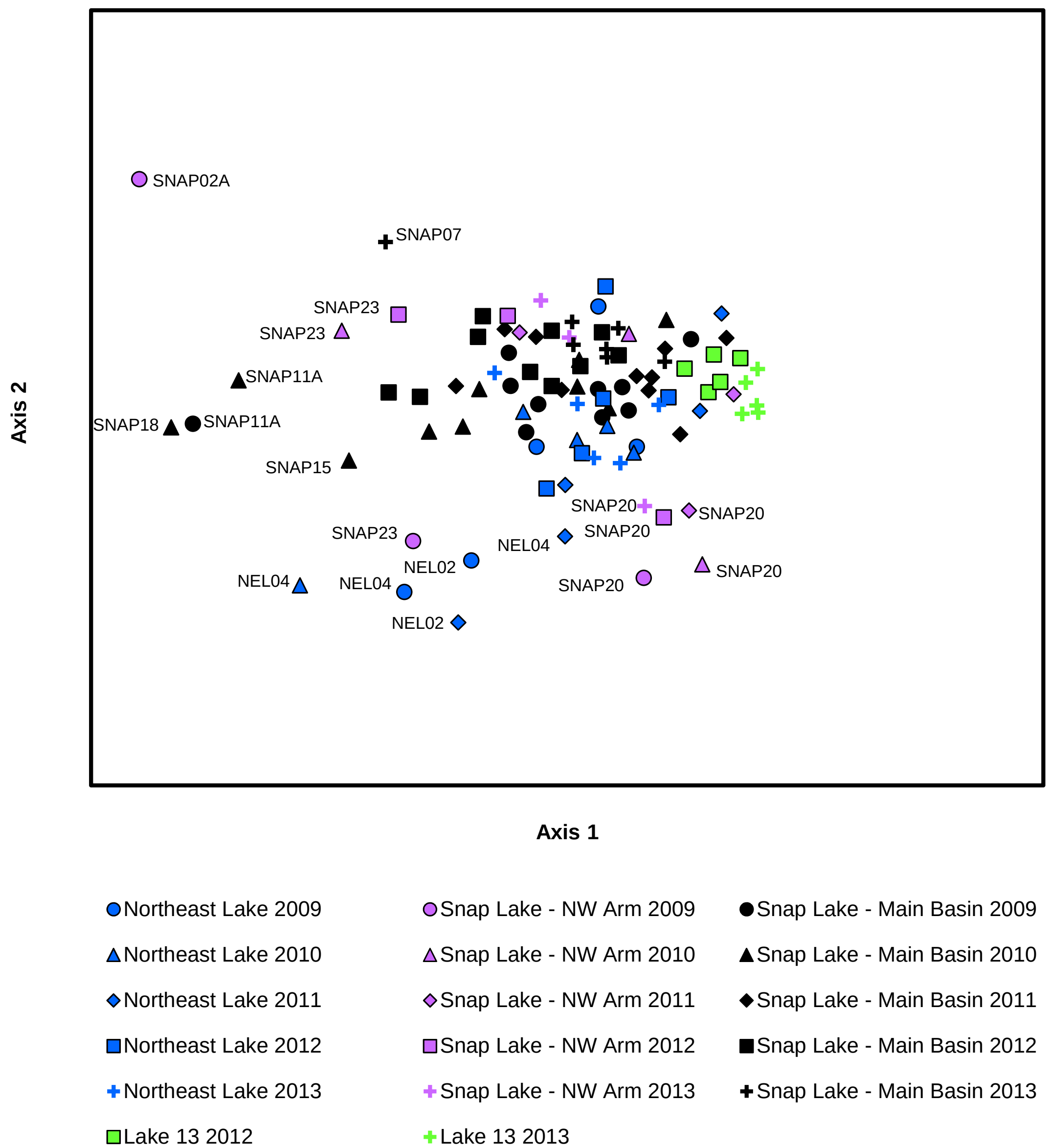


Note: Normal range represents mean ± 2 standard deviations based on Northeast Lake station means from 2009 to 2013. Error bars represent ± 1 standard error of the mean.
no./m^2 = number per square metre.

6.4.7 Multivariate Analysis

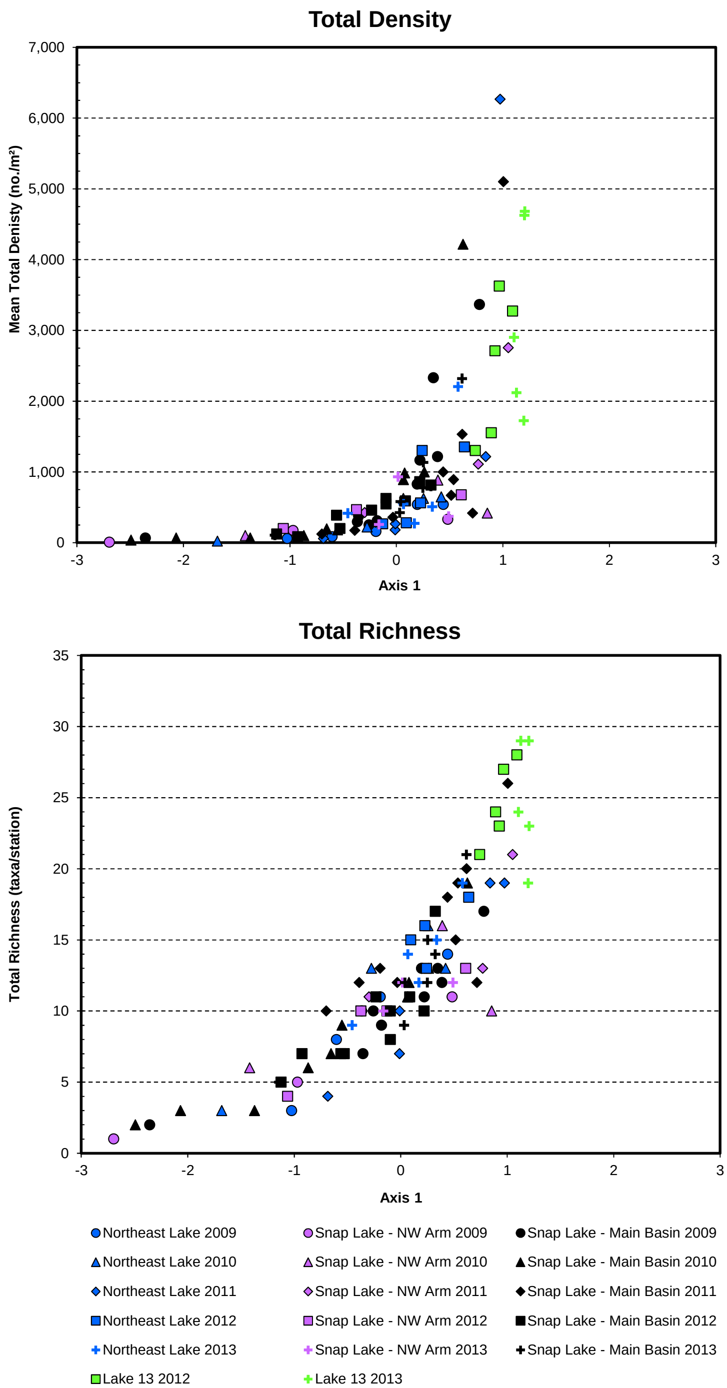
The two-dimensional configuration produced by NMDS on the 2009 to 2011 benthic invertebrate data sets had a stress value of 0.19, indicating a “fair” fit of the results to the input data, according to stress categories provided by Clarke (1993). The ordination plot of Axis 1 versus Axis 2 scores is shown in Figure 6-15. Each symbol on this figure represents the benthic community of a sampling station; stations with more similar communities are located close to one another. Relationships between Axis 1 scores, total density, and total richness are shown in Figure 6-16, and indicate a progression from communities characterized by low density and richness on the left side of Figure 6-16 to richer and denser communities on the right side of the figure.

Figure 6-15 Nonmetric Multidimensional Scaling Ordination Plot of the Fall 2009 to Fall 2012 Benthic Invertebrate Data



NW Arm = northwest arm.

Figure 6-16 Relationships between Total Density and Total Richness, and Nonmetric Multidimensional Scaling Axis 1 Scores



no./m^2 = number per square metre; NW arm = northwest arm.

The Axis 1 vs. Axis 2 ordination plot showed separation of exposure stations in Snap Lake from reference stations in Northeast Lake and Lake 13 (Figure 6-15), indicating potential evidence of a Mine-related effect. Four stations had lower scores on Axis 1 compared to reference stations. These were SNAP11A in 2009 and 2010, and SNAP18 in 2010, which are at the eastern end of the main basin; and SNAP02A in 2009, which is in the northwest arm. Other stations subject to similar exposure to the treated effluent overlapped with the range of reference station scores on Axis 1. The range of exposure station scores on Axis 2 was similar to the range of reference station scores for all years, with the exception of Station SNAP02A in 2009 and SNAP07 in 2013.

Lake 13 stations were located at the high end of the scale along Axis 1 and clustered close together, compared to stations from other sampling areas (Figure 6-15). This also suggests that Lake 13 is not a suitable lake for direct comparisons, because its benthic community differs from those present in the main basin of Snap Lake or Northeast Lake.

Some stations were identified as having very different communities compared to all other stations:

- northwest arm Station SNAP20 in 2009 and 2010;
- northwest arm Station SNAP02A in 2009;
- northwest arm Station SNAP23 in 2009;
- main basin Station SNAP15 in 2010;
- main basin Station SNAP11A in 2009 and 2010;
- main basin Station SNAP18 in 2010;
- main basin Station SNAP07 in 2013;
- NEL04 in 2009 and 2010; and,
- NEL02 in 2009 and 2011.

The reasons for the different communities at these stations are unknown.

In summary, NMDS generally ordered stations according to density and richness along Axis 1. The analysis identified some stations with unique communities. While the NMDS did not identify clusters of stations with similar communities at similar exposure to the treated effluent, some separation of exposure stations from reference stations was observed, suggesting the beginning of changes to benthic community structure associated with Mine discharges. This analysis also indicated that Lake 13 is not a suitable reference lake for direct comparison with Snap Lake to evaluate potential Mine effects on the benthic community.

6.4.8 Action Levels Assessment

No action levels were triggered for the benthic invertebrate component of the 2013 Snap Lake AEMP.

6.5 Discussion

6.5.1 Supporting Information

The 2013 Snap Lake benthic invertebrate community program represents the ninth year of monitoring after the discharge of treated effluent began and the seventh year after installation of the permanent diffuser. The 2013 results provide an opportunity to evaluate the effects of the discharge, as well as the appropriateness of the study design and reference lakes.

Water quality monitoring during winter 2005 to 2013 did not detect an effect of sufficient magnitude in DO to result in benthic community alteration (Section 3). Changes in water quality observed in deep areas of the lake included increases above the Snap Lake baseline normal range and reference lake concentrations for total alkalinity, total dissolved solids (TDS), reactive silica, total hardness, major ions (eight parameters), nitrogen parameters, and eight trace metals. In general, whole-lake means of water quality parameters were below benchmarks used in the EAR (De Beers 2002) and 2013 model predictions, with the exception of barium and uranium, which were above 2013 model predictions, likely due to model uncertainties. The whole-lake mean for antimony was well above the 2013 model prediction; however, Snap Lake concentrations of antimony have not increased and are similar to reference lakes. The concentration of TDS remained below the whole-lake average limit of 350 mg/L specified in the Water Licence (MVLWB 2013). A shift in major ions has occurred in Snap Lake. During baseline sampling the major ions in Snap Lake were calcium and bicarbonate. The relative proportion of the bicarbonate anion has decreased, while the relative proportion of the chloride anion has increased, resulting in the major ion composition in Snap Lake shifting to closely reflect the ionic composition of the treated effluent. Changes observed in sediment quality were not large enough to result in effects on the benthic community (Section 4).

Changes in water quality may influence the benthic community indirectly through altered plankton biomass. Total phytoplankton biomass increased from 2004 to 2009 followed by declines from 2009 to 2013. Annual shifts in phytoplankton and zooplankton community structure have been observed in Snap Lake over time, but the level of lake productivity has not changed substantially (Section 5) and changes in sediment TOC have not been observed. Changes in phytoplankton biomass are unlikely to influence the benthic invertebrate community through changes in settling of organic material on the lake bottom, because sediment TOC levels are naturally high in Snap Lake. Therefore, a substantial change in food availability in the form of additional organic material would be unlikely.

6.5.2 Among Lake Comparisons

The benthic community of Snap Lake in fall 2013 was characterized by variable but low total density, low to moderate richness, and dominance by Chironomidae and Pisidiidae. This type of community is expected in the sub-Arctic region where Northeast Lake, Lake 13, and Snap Lake are located (Beaty et al. 2006; Northington et al. 2010). Richness and diversity varied moderately, while evenness and density variables were highly variable. Biomass was low and highly variable among stations, and was positively correlated with total density. Lake 13 had higher total invertebrate density and richness compared to all stations sampled in both Snap Lake and Northeast Lake. Diversity was similar among all lakes; evenness was higher in Snap Lake compared to Northeast Lake and similar to Lake 13 in 2013. Differences between Northeast Lake, Lake 13, and the main basin of Snap Lake during fall 2013 in terms of taxa present were minor and not indicative of an adverse effect on the benthic community.

Statistical tests comparing benthic community variables among Northeast Lake, Lake 13, and the main basin of Snap Lake detected significant differences for the overall comparisons for total density, total richness, diversity, and the majority of densities of common taxa in 2013. Planned comparisons determined that total richness, *Micropsectra* density, and *Procladius* density were significantly lower in the main basin of Snap Lake compared to the pooled reference lakes (Northeast Lake and Lake 13 stations combined). However, statistically significant differences were detected using planned comparisons between Northeast Lake and Lake 13. Total density, total richness, diversity, Pisidiidae density (after the removal of an outlier), *Valvata sincera* density, *Procladius* density, and *Tanytarsus* density were significantly lower in Northeast Lake compared to Lake 13. This indicates that the majority of differences in statistical comparisons of the pooled references lake data to the main basin of Snap Lake are between Northeast Lake and Lake 13.

Due to the differences in the benthic invertebrate community in Lake 13, the main basin of Snap Lake was also compared directly to Northeast Lake only (i.e., Lake 13 data excluded). Using this comparison, *Micropsectra* density was significantly lower in the main basin of Snap Lake compared to Northeast Lake. *Valvata sincera* density and *Tanytarsus* density were significantly higher in the main basin of Snap Lake compared to Northeast Lake. This comparison indicates the effects on the benthic invertebrate community in the main basin are limited, and indicative of nutrient enrichment effect.

Although among-area statistical comparisons between Northeast Lake and the main basin of Snap Lake provided limited evidence of effects on the benthic community of Snap Lake, visual evaluation of the differences in abundances of dominant taxa suggests a potential Mine-related enrichment effect, which peaked from 2009 to 2011. Following this peak, total density, richness, and densities of dominant taxa declined in 2012 and increased again in 2013. Higher total density and densities of the dominant taxa (Pisidiidae, *Microtendipes*, *Valvata*, and *Procladius*), in the main basin in previous years suggest that nutrient enrichment is occurring in this area. However, 2013, total density and densities for *Microtendipes* and *Procladius* were similar between Northeast Lake and the main basin of Snap Lake. Pisidiidae and *Valvata* densities were still higher in the main basin of Snap Lake compared to Northeast Lake in 2013, indicating the enrichment is still occurring.

While the NMDS did not identify clusters of stations with similar communities at similar levels of exposure to the treated effluent, some separation of exposure stations from reference stations was observed. These results also suggest that Mine discharge may have begun to affect benthic community structure, likely due to changes in TDS and major ions in the main basin of Snap Lake.

6.5.3 Trends over Time

Trends over time were variable among lakes for benthic invertebrate summary variables. Pisidiidae density was the only variable that had values that extended above the normal range based on 2009 to 2013 Northeast Lake data. Pisidiidae density was above the normal range in all years from 2009 to 2013, with the exception of 2012. The trends comparisons among lakes for Pisidiidae density suggest a nutrient enrichment effect in the main basin of Snap Lake. The difference in trends among lakes for evenness indicates a potential Mine related effect, but due to the nature of this variable we cannot differentiate between an enrichment effect and a toxicity effect. Differences in trends among lakes for other variables do not indicate a Mine-related effect. Overall, mild nutrient enrichment appears to be occurring in the main basin of Snap Lake.

6.5.4 Reference Lakes

The benthic invertebrate community of Lake 13 is different from both Snap Lake and Northeast Lake, based on the 2012 and 2013 sampling results. Total density, richness, Pisidiidae density, *Micropsectra* density, *Heterotrissocladius* density, *Valvata sincera* density, *Procladius* density, and *Tanytarsus* density were higher in Lake 13 compared to Northeast Lake and the main basin of Snap Lake in 2013. *Heterotrissocladius* was only present in Lake 13 in 2013 All of the observed differences among lakes indicate that Lake 13 supports a richer and more abundant benthic community than Snap Lake and Northeast Lake. Lake 13 stations also clustered close together in the NMDS analysis compared to the range of stations for Northeast Lake and Snap Lake. Nutrient concentrations vary little among these lakes, with the exception of increased nitrogen concentrations in Snap Lake, which reflects inputs from the Mine. Physical factors (lake size and bathymetry) are also similar among these lakes. In the absence of chemical or physical factors, sediment characteristics may at least partly account for the variation in benthic community characteristics among lakes.

Habitat variation may be a factor in accounting for the differences in the benthic invertebrate community among lakes. Total organic carbon was lower at all stations in Lake 13 compared to Northeast Lake and Snap Lake stations. The higher sediment TOC concentrations in Northeast Lake and Snap Lake may create anoxia at the sediment water interface, which may cause the lower density and richness observed in these lakes compared to Lake 13. Total organic carbon was significantly negatively correlated with total density, richness, diversity, *Micropsectra* density, *Heterotrissocladius* density, and *Procladius* density, and significantly positively correlated with *Corynocera* density in 2013. Relationships between benthic invertebrate community variables and TOC were driven by Lake 13 stations having higher densities and lower TOC compared to Snap Lake and Northeast Lake, with no clear negative relationship present within each lake. The higher densities in Lake 13 may result from the lower TOC, which in turn may result in less anoxia at the sediment water interface compared to Snap Lake and Northeast Lake.

The statistical comparisons among lakes indicate that Lake 13 is not a suitable reference lake for direct comparisons to the main basin of Snap Lake. Total density, total richness, diversity, Pisidiidae density, *Valvata sincera* density, *Procladius* density, and *Tanytarsus* density were higher in Lake 13 compared to Northeast Lake, the established reference lake. Lake 13 is still a useful reference lake for the trends over time evaluation to determine whether the main basin of Snap Lake is diverging from lakes in the region. As an additional consequence of Lake 13 having a benthic invertebrate community variables that differs from both the main basin of Snap Lake and Northeast Lake, the normal range is based on Northeast Lake data from fall 2009 to fall 2013. This is a conservative approach because the addition of Lake 13 data increases the normal range to the point where the upper range is so high that and enrichment effect would be unlikely to be detected in the main basin of Snap Lake.

Continued use of Northeast Lake data is recommended for estimating the likely normal ranges for benthic community variables in Snap Lake. Northeast Lake remains an appropriate reference lake and it will continue to be used for direct comparisons to the main basin of Snap Lake.

6.5.5 Summary

The overall magnitude of the effect on the benthic invertebrate community in 2013 can be described as low because, although some potential Mine-related changes were detected, statistically significant differences were not found between Northeast Lake and the main basin of Snap Lake in total density or richness. The only statistically significant differences detected between Northeast Lake and the main basin of Snap Lake were for *Micropsectra* density, *Valvata sincera* density, and *Tanytarsus* density. Benthic invertebrate summary variables were within the normal ranges determined based on data from 2009 to 2013 from Northeast Lake, with the exception of Pisidiidae density in the main basin of Snap Lake. Taxonomic composition of the community has not changed appreciably compared to baseline conditions. The observed effect in 2013 is consistent with EAR predictions of a negligible to low effect on the benthic invertebrate community in Snap Lake. No action levels were triggered for the benthic invertebrate component of the 2013 AEMP.

6.6 Conclusions

6.6.1 Key Question 1: In 2013, was the benthic invertebrate community affected by the changes in water and sediment quality in Snap Lake?

Monitoring in fall 2013 detected a low level effect on the benthic invertebrate community of Snap Lake, shown as differences among lakes for some benthic invertebrate community variables, some differences in trends over time among lakes and differences in community composition. Previous years of sampling suggested a nutrient enrichment effect from the treated effluent discharge. However, differences in trends over time in the benthic invertebrate community between Northeast Lake and Snap Lake suggest that contributions from other changes in water quality, such as increasing TDS and major ions, may be influencing the benthic invertebrate community.

6.6.2 Key Question 2: If the benthic invertebrate community was affected, was the change greater than that stated in the Environmental Assessment Report?

The EAR predicted effects of negligible to low magnitude on the benthic community from construction and operation of the Mine, due to nutrient enrichment and increasing TDS concentration. The effect observed on the benthic community in 2013 was of low magnitude and is consistent with EAR predictions.

6.7 Recommendations

Results of the fall 2013 benthic survey and conductivity data collected in Snap Lake in late winter and fall 2013 were examined to recommend adjustments to the study design and data evaluation for future monitoring under the AEMP. The following recommendations are made for the AEMP benthic invertebrate program for 2015:

- Lake 13 should only be used for comparisons of trends over time with the main basin of Snap Lake. Differences in the benthic invertebrate community in Lake 13 compared to both Northeast Lake and the main basin of Snap Lake render it unsuitable for direct comparisons to the main basin of Snap Lake.
- Lake 13 data should be excluded from the calculation of the normal range for comparison to the main basin of Snap lake, because its inclusion would increase the upper limit of the normal range, reducing the potential to detect an enrichment effect in Snap Lake. Northeast Lake data from fall 2009 onward should continue to be used for estimating the normal range.
- Effects on the main basin of Snap Lake should be evaluated by comparing Northeast Lake to Snap Lake and evaluating trends over time in reference lakes to those in the main basin of Snap Lake.
- Composite samples, consisting of six individually sieved Ekman grabs combined into a single sample, should be collected at all stations beginning with the next benthic sampling program. Previous data indicate that six replicates at a station are sufficient to capture within station variability.
- Station SNAP07 should be excluded from the calculation of summary statistics for benthic invertebrate variables and statistical comparisons between Northeast Lake and the main basin of Snap Lake. SNAP07 is located near-shore in the northeast arm compared to other stations in the main basin of Snap Lake, which are in the open water. Also, it is at the shallow end of the depth range required for benthic invertebrate stations. These two factors may have contributed to the different benthic invertebrate community observed at this station in 2013.

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