DE BEERS
CANADA
SNAP LAKE MINE

May 1, 2013

File: S-115

Willard Hagen, Chair
Mackenzie Valley Land and Water Board
PO Box 2130
Yellowknife, Northwest Territories
X1A 2P6

Dear: Mr. Hagen:

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

By way of this letter, De Beers Canada Inc. (De Beers) is submitting the 2012 Aquatic Effects Monitoring Program (AEMP) to the Mackenzie Valley Land and Water Board (the Board) under Part G, Item 7 of the Water License MV2011L2-0004.

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

Sincerely,

DE BEERS CANADA INC.

A. Hood

Alexandra Hood
Permitting and Environmental Superintendant
Snap Lake Mine

Attachments

Copied to:	M. Casas	MVLWB
	D. White	SLEMA
	P. Kramers	AANDC
	B. Hanna	DFO
	D. Raymond	DBCI

EXECUTIVE SUMMARY

Introduction

De Beers Canada Inc. (De Beers) owns and operates the Snap Lake Mine (the Mine), a diamond mine located approximately 220 kilometres northeast of Yellowknife, Northwest Territories. 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 March 2013. This document represents the ninth annual AEMP report for the Mine and presents the results of the 2012 program. This is the final annual report under the July 2005 design. Future annual reports will be based on the 2013 AEMP Design Plan.

The core of the AEMP is monitoring of water quality, plankton, sediment quality, benthic invertebrates, and fish health. All monitoring components, with the exception of fish health, are currently undertaken annually. Fish health monitoring occurs on a three- to five-year cycle. It began in 2004 and was included as a component of the 2012 AEMP. The fish tasting component conducted in 2012 is included in this Annual AEMP Report. Special studies conducted in 2012 were the Littoral Zone Special Study, Downstream Lakes Special Study, Reference Lake 13 Suitability Special Study, and Nutrient Special Study.

The primary study area for monitoring in 2012 was Snap Lake. The MVLWB approved Northeast Lake as the reference lake for the AEMP in April 2006; accordingly, monitoring in Northeast Lake has been integrated into the Snap Lake AEMP. In 2012, information on a possible second reference lake (Lake 13) was collected and included.

Site Characterization and Supporting Environmental Variables

The Site Characterization and Supporting Environmental Variables component (Section 2) is a new Snap Lake AEMP component that provides and summarizes information on Snap Lake and regional aquatic environments. It provides key findings from the Surveillance Network Program (SNP) Annual Report, the Annual Air Quality and Meteorological Report, and the Hydrology Annual Report. Site information related to spills and project description changes are included as provided by De Beers' site staff. Spills from the Mine occurred, but did not adversely affect the water quality of Snap Lake.

In 2012, 27% more treated mine water was discharged to Snap Lake than in 2011. De Beers increased the volume of treated effluent released to Snap Lake during the 2012 spring freshet. A temporary floating diffuser was constructed and placed on the ice, directly above the permanent diffuser.

In 2012 rain and snowfall at Snap Lake were relatively low, but within the normal range. Air temperatures were similar to the past five years, with the exception of April, November, and December, when temperatures were lower than the long-term average.

The water surface elevation of Snap Lake increased between 2011 and 2012, but remained within the range measured from 2002 to 2011 and varied less than the three reference lakes. Peak freshet during 2012 occurred on May 18. Snap Lake inflows and outflows were within historic norms.

Temperature loggers were installed in Snap Lake and two reference lakes for the first time in 2012. The loggers indicated that Northeast Lake had colder water temperatures in the spring than the other two lakes. From July to September, the water temperatures of Snap Lake and the reference lakes were similar in shallow areas.

Based on a review of five years of ice thickness data, there was no difference between Snap Lake and Northeast Lake in terms of average annual ice thickness. Snap Lake had 226 days of ice cover in 2012, similar to the past five years.

Water Quality

The water quality component (Section 3) summarizes all data obtained from water samples and field measurements collected from Snap Lake in 2012. Over 200 water samples were collected from Snap Lake and surrounding waterbodies (i.e., Northeast Lake, Inland Lakes, Streams S1 and S27, and upstream of King Lake). In addition, water samples were collected for three special studies: the Downstream Lakes Special Study (Section 12.2); the Reference Lake 13 Suitability Special Study (Section 12.3); and, the Nutrient Special Study (Section 12.4).

Samples were shipped to analytical laboratories across Canada to obtain the best chemical analyses available. The water quality results were compared to regulatory guidelines, other benchmarks, environmental assessment predictions, and data from previous years. Water quality results from Snap Lake and the 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 effluent discharge began, with consequent increased loadings to the lake. In 2012, the annual treated effluent volume was approximately 12% of the volume of Snap Lake.

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, and some metals have increased in most areas of Snap Lake related to treated effluent discharged from the Mine. Concentrations of nitrate, chloride, and fluoride were above an AEMP benchmark (i.e., above concentrations of possible concern) on at least one occasion in 2012. Increases in these parameters were accompanied by increased hardness, which is a parameter that reduces the toxicity of those parameters. Treated effluent and receiving waters were not toxic based on laboratory toxicity testing.

Concentrations of most water quality parameters in Snap Lake were below drinking water guidelines, with the exception of *Escherichia coli* (*E. coli*), total coliforms, and possibly the metalloid antimony near the diffuser under ice. 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), so treated drinking water quality was acceptable from a microbiological perspective (*E. coli* and coliforms). The antimony results are suspect; there are indications of either contamination or analytical interference, which are being investigated to prevent this problem recurring in future sampling and analysis. In any case, antimony concentrations near the water intake were well below the drinking water guideline. 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 background concentrations within 44 kilometres (km) downstream of Snap Lake. In 2012, concentrations of Mine-related parameters reached background concentrations approximately 6 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 2012 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 is monitored annually in Snap Lake and a reference lake, Northeast Lake (Section 4). Lake 13 was monitored in 2013 as a provisional second reference lake. Sediments were collected in 2012 from 18 stations in Snap Lake, five stations in Northeast Lake, and five stations in Lake 13, and analyzed for particle size distribution, total organic carbon (TOC) content, nutrients, and metals. Average concentrations of a number of metals were higher in Northeast Lake and/or Lake 13 than in Snap Lake, indicating that concentrations of some metals are naturally elevated in this geographic region.

Overall, evaluation of trends over space and time in sediment quality did not provide clear evidence of an effect on Snap Lake sediments in areas exposed to treated effluent from the Mine. If potential effects to sediment quality have occurred to date, they have been subtle and not clearly different than natural variability. Thus, they are unlikely to have resulted in adverse environmental effects.

Plankton

The plankton component of the AEMP (Section 5) evaluated whether there were any changes happening in the small plants (phytoplankton) and animals (zooplankton) in Snap Lake waters due to nutrients or other substances added by the Mine. These small plants and animals are together referred to as plankton. Changes in plankton can affect fish in the lake since plankton are part of the food chain upon which fish rely. Such changes can potentially change the numbers and types of fish in the lake.

In 2012, plankton were evaluated at 10 locations in Snap Lake (five in the main part of the lake where the Mine is located and five in the northwest arm), once in each of July, August, and September. Plankton were also evaluated at five locations in Northeast Lake, a reference lake that is not affected by the Mine. In addition, one sample was collected in August from Lake 13, to assess whether this lake could be used as a second reference lake for plankton. Water was also collected to look at the type and amounts of nutrients (nitrogen and phosphorus) and other substances that were in the lake waters.

Concentrations of some nutrients, total dissolved solids, and chloride 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 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, Lake 13, and Northeast Lake in 2012, so nutrients or other substances released by the Mine have not had a large effect on the amount of small plants. However, the different types of small plants in Snap Lake have changed since 2004. The small animals in Snap Lake did not show an increase in numbers from 2004 to 2012, but small changes in the different types of small animals are happening in Snap Lake. These small changes are not expected to adversely affect the food chain in Snap Lake upon which fish rely.

Benthic Invertebrate 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, which form a community. They 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 2012 from 13 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. There were few differences between the established reference lake (Northeast Lake) and Snap Lake that could have been caused by the Mine discharge. Richness, which is the number of different types of benthic animals present, was lower in Snap Lake compared to Northeast Lake, and the number of snails was higher in Snap Lake compared to Northeast Lake. The benthic community also changes over time naturally and the changes observed to date are not extreme.

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. 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

The fish health component (Section 7) of the AEMP evaluated whether fish health was affected by changes in water or sediment quality in Snap Lake, and whether any observed changes were greater than those predicted in the Mine's initial environmental assessment.

Field activities in 2012 included lethal sampling of Lake Chub in Snap Lake, Northeast Lake, and the new additional reference lake, Lake 13. Each lake was sampled in early July, shortly after ice-out. Fish collected were analyzed for length, weight, age, reproductive organ weight, liver weight, female egg weight, and number of eggs per female fish. A subset of samples was analyzed for subtle changes in metabolism in the liver. Reproductive organs were also examined to confirm reproductive status, maturity, and sex. These measurements were used to estimate fish survival, energy storage, and energy use.

Although there were statistically significant differences in fish health parameters between Snap Lake and the combined reference lakes, Northeast Lake and Lake 13, these differences were within the range of natural variability and thus not biologically significant. In general, fish from both lakes were in good overall health; all indications are that fish can continue to successfully survive, grow, and reproduce in Snap Lake.

Fish Community

This section (Section 8) was not required as part of the 2012 AEMP; the Snap Lake fish community will next be sampled in 2013. This section is maintained as a placeholder for reporting in the 2013 Annual Report as detailed in Section 1.5.

Fish Tissue Chemistry

Small-bodied fish tissue chemistry (Section 9) was added to the AEMP for the first time in 2012. Previously only large-bodied fish had been collected for tissue chemistry analyses.

Eight Lake Chub carcasses (the whole body including flesh and bones, minus the liver, gonad, and stomach) were collected for tissue chemistry analyses from Snap Lake, Northeast Lake, and Lake 13. The carcasses were from the fish health survey.

Only strontium and thallium in the carcasses were significantly different (higher) between Snap Lake and the reference lakes. The difference in strontium concentration was within the range of natural variability. The difference in thallium concentration was greater than the normal range. However, there was no evidence of impaired fish health.

Fish Tasting

Fish tasting (Section 10) 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 2012, two fish were captured, prepared, and evaluated. Overall, aboriginal community members agreed that the health, taste, and texture of the fish from Snap Lake ranged from good to very good.

Traditional Knowledge

This section (Section 11) was not required as part of the 2012 AEMP, but is maintained as a placeholder for reporting in the 2013 Annual Report as detailed in Section 1.5.

Littoral Zone Special Study

A Littoral Zone Special Study (Section 12.1) was initiated in 2012 to determine the best way to sample the near-shore areas of Snap Lake and Northeast Lake. This study will continue for two more years (2013 and 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 has many different places for small plants attached to rocks (algae), animals without backbones (invertebrates), 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.

The littoral nutrient data for both Snap Lake and Northeast Lake showed that the algae lack phosphorus. Low phosphorus concentration means poorer food quality for littoral invertebrates. Food quality was poorer in Northeast Lake compared to Snap Lake, and nutrient concentrations in the littoral zone of Snap Lake were higher in 2012 compared to 2004 when a preliminary assessment was conducted before mining started. This may mean more food is available for invertebrates and fish in Snap Lake because of the nutrients discharged from the Mine. The amount of algae was higher in the littoral zone of the main basin of Snap Lake in 2012 compared to 2004, and higher in the littoral zone of Snap Lake compared to the littoral zone of Northeast Lake. The types of algae differed between 2004 and 2012 in Snap Lake, and also between Snap

Lake and Northeast Lake. Similarly, the types of littoral invertebrates collected in 2012 were different between Snap Lake and Northeast Lake. The 2012 special study showed that littoral zone monitoring is possible in Snap Lake and Northeast Lake, but the sampling method will be adjusted to collect more conclusive data. The full three years of this special study are needed to assess natural variability and the environmental significance of these differences.

Downstream Lakes Special Study

Treated effluent discharge has increased since 2004 when the Mine started operations, 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 in the initial environmental assessment, is now present in lakes downstream of Snap Lake. Results from an initial reconnaissance program in 2011 showed evidence of treated effluent (i.e., elevated dissolved salts and nutrients) throughout the first two small lakes immediately downstream of Snap Lake and within 50 metres of the inlet of Lac Capot Blanc, the third downstream lake. Water quality returned to background levels in Lac Capot Blanc.

The Downstream Lakes Special Study (Section 12.2) was conducted in August 2012 to collect further information on water quality as well as information on bathymetry, sediment and the animals living in the sediment, and plankton from the first three lakes downstream of Snap Lake. It also documented the extent of treated effluent downstream of Snap Lake.

In 2012, treated effluent was evident throughout Downstream Lake 1 and Downstream Lake 2, and again near the inlet of Lac Capot Blanc. The effluent plume was observed up to 650 metres from the inlet of Lac Capot Blanc, 600 metres further downstream than in 2011. Concentrations of dissolved salts, nutrients, and metals decreased with distance downstream, as expected. Sediment quality and the benthic community living in the sediment from the three downstream lakes were comparable to that of Snap Lake and Northeast Lake.

Reference Lake 13 Suitability Special Study

De Beers proposed that Lake 13 be added to the AEMP as a second reference lake. In 2012, as part of a Special Study (Section 12.3), De Beers collected water, sediment, plankton, benthic invertebrate, fish health, and fish tissue data from Lake 13. The objective of this Special Study was to update the baseline information on Lake 13 from 2005 to further assess the comparability of Lake 13 to Snap Lake and Northeast Lake.

Overall, water and sediment chemistry data collected in Lake 13 in 2012 were similar to data collected in 2005. There were differences in some biological components observed between the two reference lakes, Northeast Lake and Lake 13, and between Snap Lake pre-mining and Lake 13. However, the physical characteristics of the three lakes are comparable, and it is those characteristics that typically carry the heaviest weight during decisions regarding reference lake selection. Data from Lake 13 are expected to provide information on the range of natural variability within the region where Snap Lake is located.

Nutrient Special Study

The 2012 Nutrient Special Study (Section 12.4) was designed as a follow-up study to a previous nutrient study completed in 2011. A review of the nutrient data collected between 2008 and 2011 found inconsistencies in the nitrogen and phosphorus results from analytical laboratory analyses for these substances between the water quality and plankton components of the AEMP. The laboratories and sampling depths differ between the water quality and plankton programs; therefore, the cause of the inconsistent nutrient results was not clear. The 2012 Nutrient Special Study was designed to help identify the likely cause, or causes, of these differences.

Spike samples, which are samples of known concentrations, and split samples, where one sample was split into multiple samples, were used to assess the accuracy of and differences between results from the three laboratories used for nutrient analyses in the AEMP. To determine whether nutrient concentrations differed between sampling depths, nutrient results from the water quality program (i.e., mid-depth sample) and plankton program (i.e., euphotic zone, which is a sample that combines water from different depths within six metres of the water surface) were compared.

The 2012 Nutrient Special Study concluded that the accuracy of the nutrient results provided by the three laboratories was similar. However, the very low-level phosphorus concentrations in Snap Lake are difficult to analyze precisely, and result in much higher variability than spike samples with higher phosphorus concentrations. In addition, nutrient concentrations, particularly phosphorus, may differ based on sampling depth.

Qualitative Integration

The qualitative integration section of the 2012 AEMP (Section 13) combined the information and conclusions of the water quality, sediment 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 health sections. Qualitative integration was used to estimate the strength (or weight) of evidence for nutrient enrichment and 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 qualitative integration process combined laboratory determinations of nutrient (chemicals that may cause enrichment) and toxicant (chemicals that may cause toxic effects) *exposure* 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 assessed.

For 2012 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. However, there was little evidence of this nutrient enrichment transferring through the food chain to fish. 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, resulting from increases in the concentrations of some substances in water and sediment. This 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 either fish health or the structure and function of the Snap Lake ecosystem.

Action Levels

This section (Section 14) was not required as part of the 2012 AEMP, but is maintained as a placeholder for reporting in the 2013 Annual Report as detailed in Section 1.5.

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LIST OF APPENDICES[1]

[1] Appendices are numbered in order of section in which they are referred.

Acronyms

AANDC	Aboriginal Affairs and Northern Development Canada
AB	Alberta
AC	alternating current
AEMP	Aquatic Effects Monitoring Program
AITF / Alberta Innovates	Alberta Innovates Technology Futures
ALS	ALS Canada Ltd.
AML	average monthly limit
ANCOVA	analysis of covariance
ANFO	ammonium nitrate fuel oil
ANOVA	analysis of variance
AO	aesthetic objectives
APHA	American Public Health Association
AR	analytical reagent
ARD	acid/alkaline rock drainage
ARGR	Arctic Grayling
BC	British Columbia
BCMOE	British Columbia Ministry of Environment
BHP Billiton	BHP Billiton Canada Inc.
Bio-Limno	Bio-Limno Research and Consulting Inc.
BOD	biochemical oxygen demand
BTEX	benzene, toluene, ethylbenzene, and xylene
BURB	Burbot
C	carbon
Ca	calcium
$CaCO_3$	calcium carbonate
calc'd	calculated
CCME	Canadian Council of Ministers of the Environment
CCMS	collision cell inductively coupled plasma mass spectrometry
CES	critical effect sizes
CFIA	Canadian Food Inspection Agency
Chla	chlorophyll *a*
CO_3	carbonate
COC	chain-of-custody
CPUE	catch-per-unit-effort
CV	coefficient of variation
CVAAS	cold vapour atomic absorption spectroscopy
CWQG	Canadian Water Quality Guideline
D	diffuser
DC	direct current
DDW	distilled de-ionized water

De Beers	De Beers Canada Inc.
DEC	decreasing
DFO	Fisheries and Oceans Canada
DIC	dissolved inorganic carbon
DIP	dissolved inorganic phosphorus
DL	detection limit
DO	dissolved oxygen
DOC	dissolved organic carbon
DOP	dissolved organic phosphorus
DQO	data quality objective
DSL	downstream lake
DSL1	Downstream Lake 1
DSL2	Downstream Lake 2
dup	duplicate
dw	dry weight
E. coli	*Escherichia coli*
e.g.	for example
EA	Environmental Assessment
EAR	Environmental Assessment Report
EEM	Environmental Effects Monitoring
ELA	Experimental Lakes Area
EMS	Environmental Management System
et al.	and others
F	female
F1	F1 hydrocarbon fractions
F2	F2 hydrocarbon fractions
FF	Far-field
Flett	Flett Research Ltd.
GF-AAS	graphite furnace atomic absorption spectrometry
Golder	Golder Associates Ltd.
GPS	global positioning system
GSI	gonadosomatic index
H1	Hydrology Station 1
H2	Hydrology Station 2
HCO_3	bicarbonate
Hg	mercury
HG-AAS	hydride generation atomic absorption spectrometry
HVAAS	hydride vapour atomic absorption spectrometer
HydroQual	HydroQual Laboratories
i.e.	that is
i/d	immature or damaged specimen identified to the lowest level

ICP-AES	inductively coupled plasma atomic emission spectroscopy
ICP-MS	inductively coupled plasma mass spectrometry
ICP-OES	inductively coupled plasma-optical emission spectrometry
ID	identification number
IL	inland lake
IM	immature
INC	increasing
ISO	International Standards Organization
ISQG	Interim Sediment Quality Guideline
J	juvenile
K	condition factor
KING	King Lake
K-S	Kolmogorov-Smirnov
K-W	Kruskal-Wallis
LC	lethal concentration
LCB	Lac Capot Blanc
LK13	Lake 13
LKCH	Lake Chub
LKTR	Lake Trout
LNSC	Longnose Sucker
LR	lysine-arginine
LSI	liver somatic index
LSM	least squared means
M	male
m&p-Xylene	meta and para Xylene
MA	maturing
MAC	maximum acceptable concentration
Main Basin	Main Basin of Snap Lake
Max Grab	maximum allowable concentration in any grab sample
Maxxam	Maxxam Analytics Inc.
MB	Manitoba
MDS	Multi-parameter Display System
MF	Mid-field
Mg	magnesium
Mine	Snap Lake Mine
mm-dd-yyyy	month-day-year
MMER	Metal Mining Effluent Regulations Environmental Effects Monitoring Technical Guidance Document
Mo	molybdenum
MVLWB	Mackenzie Valley Land and Water Board
N	nitrogen
n	sample size

n.s.	not statistically significant
N:P	nitrogen to phosphorus
N_2	atmospheric nitrogen
NAD	North American Datum
NC	non-calculable
NE	northeast
NEL	Northeast Lake
NF	Near-field
NH_3	ammonia
NMDS	non-metric multi-dimensional scaling
NNST	Ninespine Stickleback
NO_2^-	nitrite
NO_3^-	nitrate
NRPK	Northern Pike
NW	northwest
NWT	Northwest Territories
OH	hydroxide
o-Xylene	ortho Xylene
P	probability
P	phosphorus;
PEL	Probable Effect Level
PR	pre-spawning
P-value	statistical probability
PVC	polyvinyl chloride
QA	quality assurance
QC	quality control
QS	Quick Sample
r	Pearson’s correlation co-efficient
R^2	coefficient of determination
RNWH	Round Whitefish
RPD	relative percent difference
S	sulphur
sample	field sub-sample
SCN	sample control number
SCUBA	self-contained underwater breathing apparatus
SD	standard deviation
SE	standard error
Se	selenium
Si	silica
SiO_2	silicate
SLSC	Slimy Sculpin

SM	standard method
SNAP	Snap Lake
SNP	Surveillance Network Program
Sonar/GPS	sonar coupled with a global positioning system
SQG	sediment quality guideline
SR	standardized residuals
SWI	specific work instruction
Taiga	Taiga Environmental Laboratory
TDN	total dissolved nitrogen
TDP	total dissolved phosphorus
TDS	total dissolved solids
TDS_{Calc}	calculated total dissolved solids
TDS_{meas}	measured total dissolved solids
TEH	total extractable hydrocarbons
TIP	total inorganic phosphorus
TKN	total Kjeldahl nitrogen
TN	total nitrogen
TOC	total organic carbon
TOP	total organic phosphorus
TP	total phosphorus
TS	temporary sump
TSS	total suspended solids
TVH	total volatile hydrocarbons
TWTP	temporary water treatment plant
U	unknown
UofA	University of Alberta Biogeochemical Analytical Service Laboratory
USEPA	United States Environmental Protection Agency
UTM	Universal Transverse Mercator
V	volt
WH	warning qualifier
WHO	World Health Organization
WMP	water management pond
WOE	weight of evidence
WQ	water quality
WQG	water quality guideline
wt	weight
WTP	water treatment plant
X	times
yr	year
YOY	young-of-the-year
α	alpha

β	beta
↓	decrease
↑	increase
↑/↓	rating 1
↑↑/↓↓	rating 2
↑↑↑/↓↓↓	rating 3

Units of Measure

%	percent
% dw	percent dry weight
<	less than
>	greater than
±	plus or minus
≤	less than or equal to
°	degree (angle)
°C	degrees Celsius
µg ww	micrograms wet weight
$µg/cm^2$	micrograms per square centimetre
µg/g	micrograms per gram
µg/g ww	micrograms per gram wet weight
µg/L	micrograms per litre
$µg/m^3$	micrograms per cubic metre
µm	micrometre
$µmol/cm^2$	micromoles per square centimetre
µS/cm	microSiemens per centimetre
A	amp
$cells/cm^2$	cells per square centimetre
cells/L	cells per litre
CFU/100 mL	colony forming unit per 100 millilitres
cm	centimetre
cm^2	square centimetre
g	gram
g ww	grams wet weight
h	hour
ha	hectare
kg	kilogram
kg/year	kilograms per year
km	kilometre
km^2	square kilometre

L	litre
m	metre
m^2	square metre
m^3	cubic metre
m^3/d	cubic metres per day
m^3/s	cubic metres per second
masl	metres above sea level
mg	milligram
mg/kg	milligrams per kilogram
mg/kg dw	milligrams per kilogram dry weight
mg/L	milligrams per litre
mg/m^3	milligrams per cubic metre
mg-N/L	milligrams as nitrogen per litre
mg-P/L	milligrams as phosphorus per litre
mL	millilitre
mm	millimetre
Mm^3	million cubic metres
mm^3/m^3	cubic millimetres per cubic metre
MPN/100 mL	most probable number per 100 millilitre
MΩ	megaohm
$no./m^2$	numbers per square meter
NTU	nephelometric turbidity unit
org/m^2	organism per square metre
org/m^3	organism per cubic metre
ww	wet weight

Glossary

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.
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.
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.
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.
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.
biota	Living organisms.
Boreal Forest	The northern hemisphere, circumpolar, tundra forest type consisting primarily of black spruce and white spruce with balsam fir, birch, and aspen.

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.
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.
colonial	Individuals of the same species clustered together to form a group.
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.
Copepoda	An order of planktonic crustaceans; a component of zooplankton.
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.
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.
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.
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.
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.

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.
geographic information system (GIS)	Computer software designed to develop, manage, analyze, and display spatially referenced data.
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.
hydrology	The science of water movement and distribution, including the hydrologic cycle and interactions with the physical and biological environment.
ice-covered conditions	The period of time, during the year, when waterbodies are covered in ice.
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.

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.
mesotrophic	Trophic state classification for lakes characterized by moderate productivity and nutrient inputs (particularly total phosphorus).
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).
near-field	Stations located in the north basin of Snap Lake.
normal range	An estimate of natural variability calculated as ± 2 standard deviations of the reference mean or ± 2 standard deviations of the Snap Lake baseline, as appropriate
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).
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, i.e., P <0.05.

particulate matter A mixture of small particles, e.g., dust and soil.

pelagic Open-water area within a lake.

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.

polygon Representations of an area consisting of a plane figure bounded by straight edges.

probable effect levels Concentration of a chemical in sediment above which adverse effects on an aquatic organism are likely but not certain to occur.

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.

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.

relative abundance The proportional representation of a species in a sample or a community.

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.

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.
specific conductivity	A measure of how well water conducts electricity.
spring freshet	A spring thaw event resulting from melting snow and ice.
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.
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).
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.
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.
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.
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.
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).

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 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 Environmental Impact Review Board (MVEIRB) 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 MVEIRB (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.

The Mine has been operating under the terms and conditions of a Class A Water Licence issued in 2004 (Licence #MV2001L2-0002; MVLWB 2004). 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 by the Mackenzie Valley Land and Water Board (MVLWB) for a period of eight years, effective June 14, 2012 (Licence #MV2011L2-0004; MVLWB 2012).

The Aquatic Effects Monitoring Program (AEMP) is a requirement of the Water Licence Part G (MVLWB 2012). 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 for 2012 was based on the final, approved AEMP Design Plan submitted to the MVLWB in June 2005 (De Beers 2005).

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. Aboriginal community members participated in the fish-tasting events in 2012 and in previous years (Section 10).

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 years of monitoring. The 2013 AEMP Design Plan (De Beers 2012) was submitted to the MVLWB November 2012 with the intent that, following MVLWB review and approval, it would be implemented in 2013. Where applicable, aspects of the 2013 AEMP Design Plan were included in the 2012 AEMP. In March 2013, the MVLWB approved the 2013 AEMP Design Plan with conditions.

LEGEND

- SNAP LAKE MINE
- EXISTING MINE
- TERRITORIAL CAPITAL
- POPULATED PLACE
- HIGHWAY
- EXISTING WINTER ROAD
- TIBBITT-TO-CONTWOYTO WINTER ROAD
- WATERCOURSE
- WATERBODY
- TERRITORIAL/PROVINCIAL BOUNDARY

100 0 100

SCALE 1:3,500,000 KILOMETRES

REFERENCE

Base data source: The Atlas of Canada
Datum: NAD83 Projection: UTM Zone 12

PROJECT

DE BEERS
GROUP OF COMPANIES

TITLE

LOCATION OF SNAP LAKE MINE

Golder Associates

PROJECT No.	13-1349-0001.1100	
DESIGN	TD	2013-02-27
CADD	JEF	2013-03-25
CHECK	LMB	2013-04-19
REVIEW	PMC	2013-04-19

FILE No. 13134900011100A001-1
SCALE AS SHOWN

FIGURE 1-1

L:\2013\1349\13-1349-0001\Phase 1100\Report A\13134900011100A001-1.dwg | Layout: Location of Snap Lake Mine | Modified: jfarah 04/23/2013 11:32 AM | Plotted: jfarah 04/23/2013

1.2 OBJECTIVE AND SCOPE

1.2.1 Objective

This document represents the ninth AEMP annual report for the Mine and presents the results of the 2012 program. The main objectives of the 2012 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 2012 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 stations are integrated into the AEMP and are included in this report. All Surveillance Network Program and AEMP monitoring activities are reported in the Water Licence Annual Report.

Table 1-1 Aquatic Effects Monitoring Program (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	Section 2 to Section 13 and appendices
e)	an interpretation of the results, including an evaluation of any identified environmental effects that occurred as a result of the Mine	Section 2 to 12; summarized in Section 13
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 13
g)	a comparison of monitoring results to Action Levels as set in the AEMP Design Plan	To be reported in Section 14 of the 2013 AEMP Annual Report
h)	an evaluation of the overall effectiveness of the AEMP to date	Section 13
i)	recommendations for refining the AEMP to improve its effectiveness as required	Sections 2 to 11; Section 15 Recommendations
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 12 (Special Studies)

AEMP = Aquatic Effects Monitoring Program.

1.2.2 Scope

The core 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, which are discussed in greater detail in the following sections. The 2012 AEMP monitoring components are:

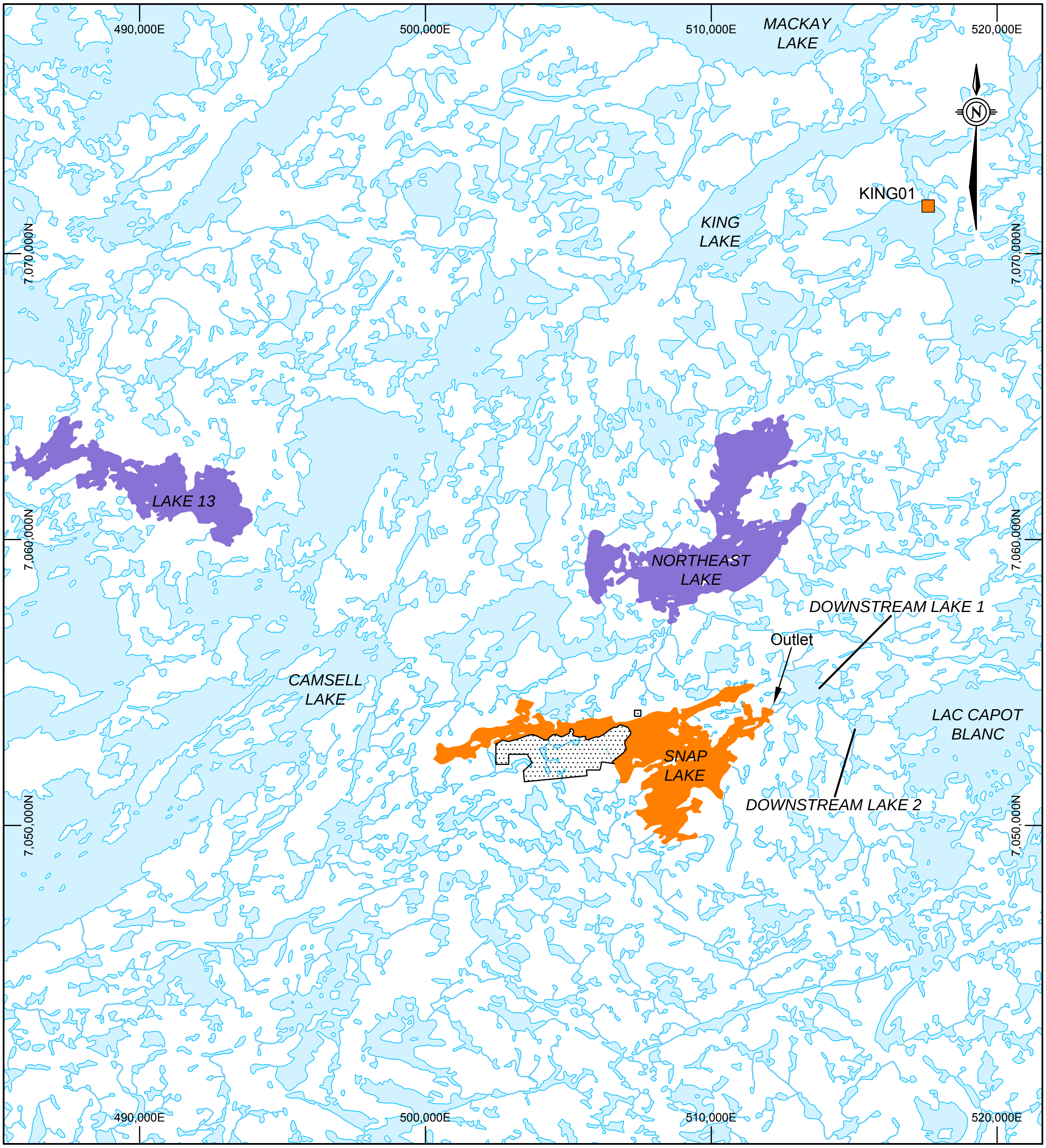
- site characterization;
- water quality;
- sediment quality;
- plankton;
- benthic invertebrate community;
- fish health;
- fish tissue chemistry; and,
- fish tasting.

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 2011 AEMP program and included in this report are: Littoral Zone Special Study (Section 12.1); Downstream Lakes Special Study (Section 12.2); Reference Lake 13 Suitability Special Study (Section 12.3); and, Nutrient Special Study (Section 12.4).

The MVLWB (2006) approved Northeast Lake as the reference lake for the AEMP. The objective of sampling a reference lake is to verify that changes associated with external factors, such as climate change, are not attributed to the Mine. Monitoring in Northeast Lake began in 2006 and was integrated into the Snap Lake AEMP. In the 2013 AEMP Design Plan (De Beers 2012), it was recommended that a second reference lake be added to the AEMP program. In March 2013, Lake 13 was approved by the MVLWB as a second reference lake. Water quality, benthic invertebrate, sediment quality, plankton, fish health, and fish tissue sampling occurred at Northeast Lake and Lake 13 in 2012, and the suitability of Lake 13 as a reference lake was included in the 2012 AEMP as a special study (Section 12.3).

1.3 STUDY AREAS

The study areas for the 2012 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 2012 that focused on three lakes immediately downstream of Snap Lake (Section 12.2).



LEGEND

WATER QUALITY STATION
MONITORING AREA
REFERENCE AREA
SNAP LAKE MINE FOOTPRINT
WATERBODY

4 0 4
SCALE KILOMETRES

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.

PROJECT

DE BEERS
GROUP OF COMPANIES

TITLE

STUDY AREA, 2012 AEMP

Golder Associates

PROJECT No.	13-1349-0001.1100		FILE No.	13134900011100A001-2
DESIGN	TD	2013-03-13	SCALE	AS SHOWN
CADD	JEF	2013-03-29		
CHECK	LMB	2013-04-19	**FIGURE 1-2**	
REVIEW	PMC	2013-04-19		

L:\2013\1349\13-1349-0001\Phase 1100\Report A\13134900011100A001-2.dwg | Layout: Fig 1-2 | Modified: jfarah 04/01/2013 10:35 AM | Plotted: jfarah 03/28/2013

1.4 SITE ACTIVITIES IN 2012

Major construction activities and milestones achieved during 2012 by De Beers were:

- construction of the East Cell embankments and ribs (1-4);
- construction of inland lake (IL) IL6 diversion ditch catchment;
- de-mobilization of temporary camp on the winter road;
- installation of temporary booster pump and pad on the diffuser line;
- expansion of apron quarry and relocation of STP;
- construction of pads for perimeter sump pump stations;
- poured concrete pad in the Utility Plant for relocation of the STP; and
- relocation of landfill to East Cell 1.

As required under Part G, Item 8b of the Water Licence (MV2011L2-0004), De Beers has reviewed site activities for 2012. Spills and leaks that occurred on site during this period were contained and mitigated. The possible influence of spills and leaks on water quality in Snap Lake, and their potential effects to organisms living in the lake are considered in this 2012 AEMP report and outlined in further detail in Section 2.

1.5 2012 REPORT ORGANIZATION

The following sections comprise this 2012 AEMP Report and are, with the exception of Section 16 (Closure), summarized in plain-language form in the Executive Summary:

- 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 7 – Fish Health;
- Section 9 – Fish Tissue Chemistry;
- Section 10 – Fish Tasting;
- Section 12 – Special Studies (Littoral Zone Special Study, Downstream Lakes Special Study, Reference Lake 13 Suitability Special Study, Nutrient Special Study);

- Section 13 – Qualitative Integration;
- Section 15 – Recommendations, and,
- Section 16 – Closure.

The following sections were not required as part of the 2012 AEMP, but will be incorporated into the 2013 Annual report as part of the new 2013 AEMP Design Plan. These sections have been maintained in this report as placeholders for future sections in the 2013 Annual Report:

- Section 8 – Fish Community;
- Section 11 – Traditional Knowledge; and,
- Section 14 – Action Levels.

The fish community component of the Snap Lake AEMP is conducted every three to five years; it was last conducted in 2009, will next be conducted in July 2013, and reported in the 2013 Annual Report. Further planning on Traditional Knowledge will occur in May 2013 and will be included in the 2013 Annual Report. The Action Levels of the 2013 AEMP Design Plan will be reviewed in May and June 2013 and will be added in the 2013 Annual Report as per the Water Licence (MV2011L2-0004) requirement (Part G, Item 8g, MVLWB 2012).

1.6 REPORT PREPARATION

De Beers retained Golder Associates Ltd. (Golder) for the design, implementation, analysis, and reporting of the AEMP, with the exception of the fish tasting report (Section 10) which was prepared directly by De Beers. De Beers chose Golder as an appropriately qualified organization with comprehensive scientific expertise in the areas of water and sediment quality, aquatic biology, and fish. This report was prepared by discipline-specific teams of professionals with appropriate scientific credentials, extensive environmental assessment and aquatic effects monitoring experience, and relevant technical skills including field sampling, data analysis and interpretation.

Golder subcontracted specialized water and sediment chemistry analyses laboratory work to; Maxxam Analytics Inc., Edmonton, Alberta (AB); Flett Research Ltd., Winnipeg, Manitoba (MB); Alberta Innovates, Vegreville, AB; and, the University of Alberta, Edmonton, AB. Toxicity testing was subcontracted to HydroQual Laboratories, Calgary, AB. Advanced Eco-Solutions Inc. (Newman Lake, Washington, United States), Eco-Logic Ltd. (Vancouver, British Columbia [BC]), and Bio-Limno Research and Consulting Inc., (Halifax, Nova Scotia) were subcontracted to conduct plankton and zooplankton taxonomy and plankton and zooplankton biomass measurements. Zloty Environmental Research and Consulting Ltd. (Summerland, BC) were subcontracted to conduct benthic invertebrate enumeration and taxonomy. North South Consultants Inc. (Winnipeg, MB: fish aging), North Carolina State University (Raleigh, North

Carolina: gonad histology), University of Saskatchewan (Saskatchewan: liver lipids and glycogen), and Zloty Environmental Research and Consulting Ltd. (Summerland, BC: stomach content determination) were subcontracted for the fish health component. ALS Laboratory Group (Burnaby, BC) was retained by De Beers to conduct water, sediment, and fish tissue chemistry analyses.

1.7 REPORT LIMITATIONS

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, express or implied, is made.

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The factual data, interpretations, suggestions, recommendations, 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. In order to properly understand the factual data, interpretations, suggestions, recommendations, and opinions expressed in this document, reference must be made to the entire document.

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1.8 REFERENCES

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2 SITE CHARACTERIZATION AND SUPPORTING ENVIRONMENTAL VARIABLES

2.1 INTRODUCTION

2.1.1 Background

Site Characterization and Supporting Environmental Variables is a new component for the 2012 Aquatic Effects Monitoring Program (AEMP). The purpose of this component is to summarize information to describe the general conditions at the Mine site and the local environment in which the AEMP is conducted. This component incorporates key information relevant to the Snap Lake aquatic environment and additional habitat data. The data presented in this Site Characterization and Supporting Environmental Variables section 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, fish health, and fish community).

2.1.2 Objectives

The primary objective of the Site Characterization and Supporting Environmental Variables component is to provide a description of the non-Mine related modifying factors that may affect the Snap Lake ecosystem, and that need to be considered during data interpretation by each AEMP component. In the Site Characterization and Supporting Environmental Variables component, the relevant data are summarized and presented; the main AEMP components subsequently consider this section during the interpretation of their own component-specific data.

Information on the characteristics of the Mine site and its operations, as well as characteristics of the surrounding waterbodies, was generally reported in the Environmental Assessment Report (EAR; De Beers 2002), and is updated in annual reports prepared outside of the AEMP:

- Surveillance Network Program annual reports;
- Hydrology Annual Report; and,
- Air Quality and Meteorological Annual Report.

An overall description of the aquatic habitat in Snap Lake is not included in the above reports; therefore, additional information on habitat was collected during 2012 as part of the Site Characterization and Supporting Environmental Variables component (i.e., seasonal water temperature, ice thickness, and duration of ice cover). Where available, the data collected at Snap Lake are compared to data from nearby reference lakes. The water temperature monitoring program compares data collected at Snap Lake to data from Northeast Lake and provisional reference Lake 13 (hereafter referred to as Lake 13), which is being assessed for possible inclusion in the AEMP (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

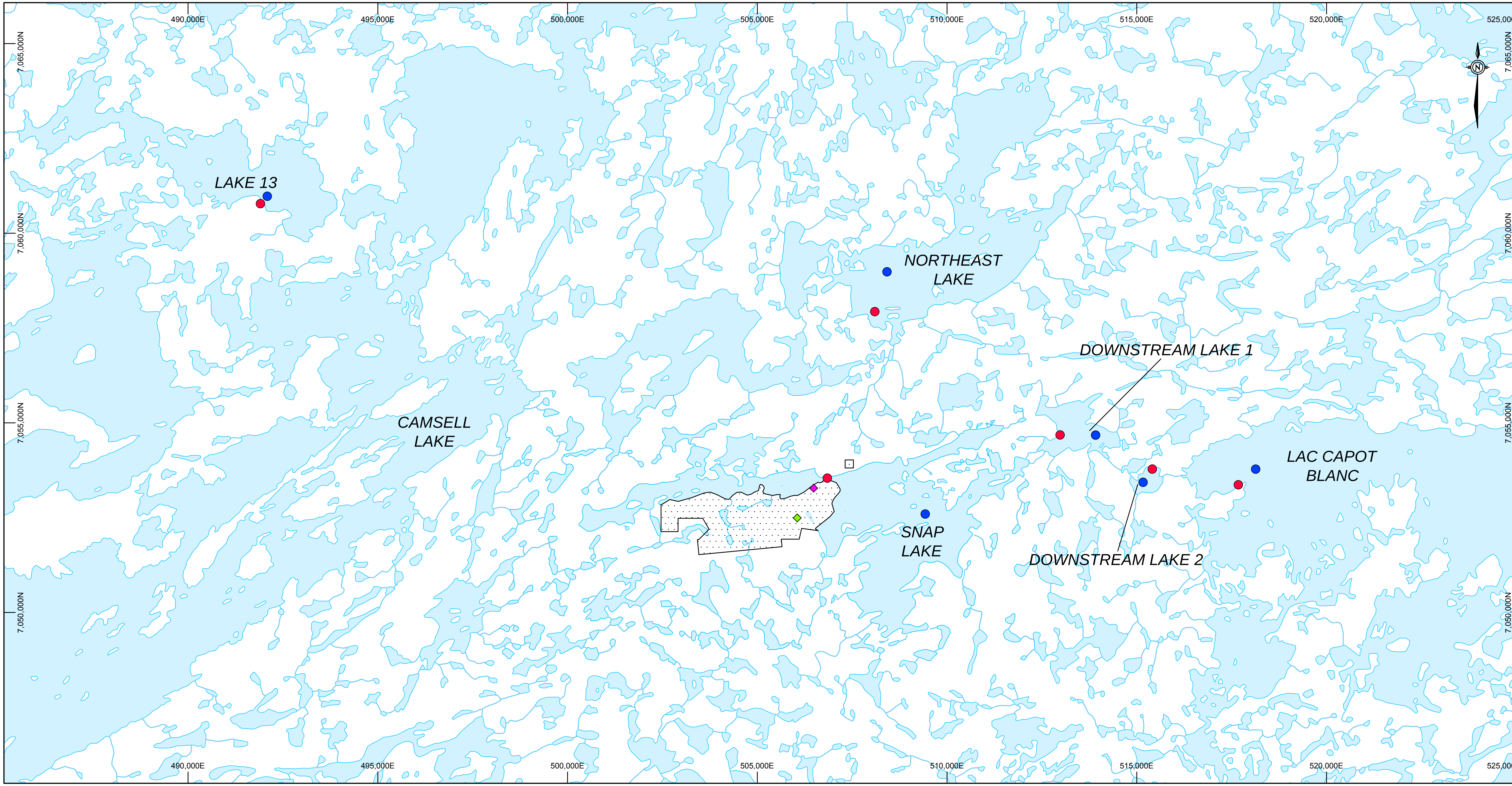
The conditions at the Mine site in relation to the aquatic environment were characterized by reviewing information provided by the De Beers Snap Lake Mine site staff.

Between December 16, 2011 and December 15, 2012, 30 reportable spill events were summarized by De Beers site staff[2]. The summary was subsequently screened for spills that occurred in a location in or near a waterbody and of a volume considered to be large enough to possibly affect the aquatic environment.

Current site conditions were compared by De Beers to the Consolidated Project Description for the Snap Lake Mine submitted to the Mackenzie Valley Land and Water Board in 2003 (De Beers 2003), and summarized. The operational changes to the project description would not be considered new information for regulators or stakeholders, but are summarized for consideration by AEMP components.

The volume of treated effluent discharged is a summary of daily flow information provided by De Beers site staff[3]. The volume of treated effluent combines daily flows to Snap Lake from the water treatment plant and the temporary water treatment plant. Daily discharge flows from the sewage treatment plant are not included. These data are used to determine monthly trends and daily discharge volumes for 2012.

[2] "Spills_2012.xlsx" provided by De Beers January 20, 2013
[3] "2012 License Reporting Requirements" provided by Be Beers January 20, 2013



SCALE 2 0 2 4 KILOMETRES

LEGEND

- TEMERATURE LOGGER LOCATION (SHALLOW)
- TEMERATURE LOGGER LOCATION (DEEP)
- METEOROLOGICAL LAKE SAMPLING STATION
- METEOROLOGICAL HILL SAMPLING STATION
- SNAP LAKE MINE FOOTPRINT
- WATERBODY

REFERENCE

DIGITAL MAP FROM MACKAY LAKE, NORTHWEST TERRITORIES, PRODUCED BY DEPARTMENT OF ENERGY, MINES AND RESOURCES. MAP 75M, ORIGINAL SCALE 1:250,000, NAD 83 UTM ZONE 12.

SNAP LAKE OUTLINE AND ISLANDS WERE CORRECTED TO LANDSAT 7 SATELLITE IMAGE 45/15, DATED SEPTEMBER 2, 2000. PROVIDED BY GEOBASE.

NOTES

1. MAP MOVED TO NAD83 COORDINATE SYSTEM.

PROJECT

DE BEERS GROUP OF COMPANIES

TITLE

TEMPERATURE LOGGER LOCATIONS, 2012

Golder Associates

PROJECT	13.1349.0001.5100	FILE No. 13134900016000A002	
DESIGN	DC	2013-01-21	SCALE AS SHOWN REV. 0
CADD	JEF	2013-02-01	**FIGURE 2-1**
CHECK	LMB	2013-04-19	
REVIEW	PMC	2013-04-19	

L:\2013\1349\13-1349-0001\Phase 6000\Report_A\ Drawing file: 13134900016000A2-1.dwg Apr 23, 2013 - 11:52am

LEGEND

- HYDROLOGICAL MONITORING LOCATION
- WATERCOURSE
- WATERBODY
- Snap Lake Mine Footprint

2 0 2

SCALE 1:125,000 KILOMETRES

NOTE

DSL1 = First lake downstream of Snap Lake.

REFERENCE

Base Data: Canvec 1:50000 downloaded from Geogratis
DATUM: NAD83 PROJECTION: UTM ZONE 12

PROJECT

DE BEERS
GROUP OF COMPANIES

TITLE

HYDROLOGICAL MONITORING LOCATIONS, 2012

Golder Associates

PROJECT		13-1349-0001	FILE No.	
DESIGN	KM	09 Apr. 2013	SCALE AS SHOWN	REV. 0
GIS	KM	09 Apr. 2013	FIGURE: 2-2	
CHECK	LMB	19 Apr. 2013		
REVIEW	PMC	19 Apr. 2013		

N:\Active\GIS\2013\Misc_Request\Snap_SurroundingLakes

2.2.2 Meteorological Monitoring

Meteorological data including rainfall, temperature, wind, relative humidity, and solar radiation are reported in the Air Quality and Meteorological Annual Report (Golder 2012a).

During 2012, meteorological data were collected at the hill meteorological monitoring station (Hill Station) located on an elevated point of land immediately west of the water management pond (WMP), and at the lake hydro-meteorological monitoring station (Lake Station) located northeast of the construction camp. Rainfall data were not collected at the Lake Station due to a hardware malfunction. The locations of the Hill Station and the Lake Station are shown on Figure 2-1.

The meteorological data were collected, reviewed, and figures produced by the authors of the Annual Air Quality and Meteorological Report (Golder 2012a), as well as the interpretation of the data in general terms. The data collected at Snap Lake are compared to the Environment Canada data collected in Yellowknife (Environment Canada 2012).

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 2012b). Benchmarks were established by setting metal pins into bedrock and surveying the pins for elevation in metres above sea level (masl). Benchmarks on Snap Lake allow for water elevation measurements at outflow (H1 and H2), lake elevation (H3), and inflow (H4) locations. North Lake, Northeast Lake, and 1999 Reference Lake each have a benchmark for lake elevation measurements.

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. Over time, the relationship between water elevation (stage) in the channel and flow (discharge) was established and described by a stage-discharge rating curve. The stage-discharge curve can subsequently be used to calculate the discharge based on the elevation of the water at the monitoring station.

The benchmark locations and elevations are shown in Table 2-1 and in Figure 2-1.

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.

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 2012b). A tag line marked at 0.2-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, the velocity was measured at 80% and 20% of depth, and the measured velocities were averaged.

The product of the mean of the depths and the mean of the velocities observed at adjacent segments was multiplied by the width between the centre points of segments to determine the discharge for each segment. This method was repeated for each consecutive segment across the stream and the total discharge (in cubic metres per second [m^3/s]) for the stream was then calculated by summing the partial discharges.

2.2.3.3 Continuous Water Level Recording

Water surface elevations were measured every 30 minutes using a Levelogger Gold 3001 (manufactured by Solinst Canada Ltd.) during the open-water period at streamflow monitoring stations H1 and H2, and year-round at Station H3. Water surface elevations at the 2005 Benchmark were surveyed at least monthly by Nampcy Solutions Ltd. using a rod and level. The water surface elevation at H4 was recorded every 30 seconds during the period of peak flow.

At the stream stations (H1 and H2), the Leveloggers are mounted to brackets that are installed in the streambed. At the Snap Lake Station (H3), the Levelogger is located in approximately 1.2 m deep water and about 5 m from shore. The H3 logger data began to drift away from the surveyed data during summer 2012; therefore, the surveyed data are used in the water balance. Golder has recommended that De Beers replace all of the loggers for the 2013 season.

2.2.4 Water Temperature Monitoring

During the AEMP field programs, water temperature data were collected at the Snap Lake Mine site in the following study lakes:

- Snap Lake;
- Northeast Lake;
- Lake 13;
- Lac Capot Blanc;
- Downstream Lake 1; and,
- Downstream Lake 2.

Three temperature loggers (TidbiT Water Temperature Data Loggers - UTBI-001), also referred to as thermographs, were installed in each of the six study lakes. The temperature loggers were installed July 10 and 11, 2012, and removed September 10 and 11, 2012. The temperature loggers were programmed to record the water temperature hourly.

One shallow site location (i.e., less than 1.0 m depth) and one deep site location (i.e., overall water depth of 10 to 15 m) were selected at each study lake. At the shallow site, one temperature logger was installed mid-depth, approximately 0.5 m below the water surface. At the deep sampling site, two temperature loggers were installed; one logger was installed 0.3 m below the water surface (deep site/surface logger), and one bottom logger was installed 1.0 m above the bottom substrate (deep site/bottom logger). The locations for temperature loggers are shown in Figure 2-1.

The deep site/surface logger at Snap Lake was lost during the retrieval field program and no data were recovered. The locations of the temperature loggers installed in Lac Capot Blanc and two downstream lakes are shown in Figure 2-1; the results for these three lakes are discussed in the Downstream Lakes component (Section 12.2).

The water temperature data were reviewed by site and depth.

2.2.5 Ice Thickness and Ice Cover Monitoring

Ice thickness measurements were collected at Snap Lake and Northeast Lake during the winter AEMP water quality field programs. The mean annual ice thickness was calculated for Snap Lake using ice thickness data from 2005 to present, and for Northeast Lake using data from 2008 to present. Annual average ice thicknesses of these two lakes were compared with a Mann-Whitney U-test. Ice thickness measurements are not available for Lake 13 because AEMP field crews have not sampled this lake when there is ice cover. Ice thickness measurements for Lake 13 will be collected in May 2013.

Days of ice cover were determined for Snap Lake from De Beers site staff observations. The first day of ice cover was considered to be the date in which a layer of ice was observed on the main basin of Snap Lake. The last day of ice cover was considered to be the day in which the main basin ice layer melted. No information on days of ice cover is available for the remote reference lakes, as daily observations from site staff would be required to obtain this information.

2.3 QUALITY ASSURANCE AND QUALITY CONTROL

2.3.1 Overview of Procedures

Quality assurance (QA) and quality control (QC) procedures are an important aspect of any field or laboratory testing program. The objective of the QA/QC program is to standardize methods so that field sampling, data entry, data analysis, and report preparation produce technically sound and scientifically defensible results.

Data presented from the air quality Hill and Lake Stations, hydrology, and Mine operations components have undergone QA/QC review as part of those components.

The QA/QC program for the water temperature logging program involved comparing logger data to field measurements collected during AEMP field programs to check for completeness, accuracy, and consistency during processing.

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

2.3.2 Summary of Results

The data collected on the first two days following installation of the water temperature loggers and on the day of their removal 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.

The Snap Lake deep station temperature logger data, installed at 1 m above the substrate, did not match AEMP field measurements and was inconsistent with temperature trends measured by other loggers. The data from this temperature logger were thus removed from the dataset analyzed as part of Section 2.4.4.

2.4 RESULTS

2.4.1 General Site Condition Monitoring

2.4.1.1 Spills

The De Beers site staff recorded 30 reportable spill events which occurred between December 16, 2011 and December 15, 2012. The complete list was reviewed to identify spills at the Snap Lake Mine site with the potential to affect the aquatic environment. Five such spill events were identified (Table 2-2). Low volume spills (i.e., less than 3 cubic metres [m^3]) are not included in this Table 2-2, as the majority were contained, did not enter a waterbody, and were not anticipated to have a measurable effect on the aquatic environment.

Table 2-2 Spills of Process Water and Treated Effluent at the Snap Lake Mine Site between December 16, 2011 and December 15, 2012

Date	Product	Volume	Location	Description
December 29, 2011	Process Water	5,500 m^3	Temporary Sump 4	Sump overflowed to the tundra due to sudden influx of water
February 29, 2012	Process Water	13 m^3	Toes of Dam 1 TH06-10	Seepage due to insulating snow effects
May 27, 2012	Treated Effluent	40 m^3	Pipebench downstream of Water Treatment Plant	Uncontrolled release of treated effluent
August 1, 2012	Treated Effluent	293 m^3	Water Treatment Plant	Discharge of non-compliant treated effluent
December 5, 2012	Process Water	12 m^3	Perimeter Sump 3	Water line came apart

Source: Spills_2012.xlsx provided by De Beers January 20, 2013.
m^3 = cubic metres.

Mitigation methods were put in place for the December 29, 2011 spill. A berm was constructed down-gradient from Temporary Sump 4 (TS4) to prevent the movement of process water. A diversion ditch leading to Inland Lake 6 was constructed prior to freshet to divert process water which had been frozen in place during the December 29, 2011 spill. Any changes to Snap Lake from this runoff inflow were localized, temporary, and negligible relative to changes from the treated effluent from the diffuser (Golder 2012c).

The August 1, 2012 incident resulted in the discharge of treated effluent through the diffuser pipe, characterized by turbidity levels exceeding 7 Nephelometric Turbidity Units (NTU) for less than 10 minutes (De Beers 2012).

2.4.1.2 Project Description Changes

The Consolidated Project Description for the Snap Lake Mine was submitted to the Mackenzie Valley Land and Water Board in 2003 (De Beers 2003). Table 2-3 summarizes the operational changes that have occurred since the original project description.

Table 2-3 Project Description Changes

Topic	Original Project Description	Current Site Conditions
Freshwater	Freshwater will be drawn from Snap Lake to the Process Plant and drill water for underground drilling.	No freshwater is used in the Process Plant. Freshwater is drawn from Snap Lake only 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. Traps will collect sediment generated from outlying areas.	Rockhill ditches and grading direct runoff from the peninsula areas to the WMP. Traps are now referred to as sumps.
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 land-farm will be pumped to the WTP.	Runoff water is pumped to the WMP, not directly to the WTP. Ditches now referred to as sumps.
Dust suppression	Water for dust suppression of the North Pile will be drawn from the WTP at a rate of 55 m^3/d for six months per year.	The North Pile is not sprayed for dust suppression.
Dam raises	Dam 1 and Dam 2 will be raised by 2 m each to increase capacity.	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 is sent to the WMP.
Sewage treatment	The sludge from the sewage treatment plant will be incinerated and placed in the landfill.	The sludge from the sewage treatment plant will be incinerated or placed in the landfill.
Methods of transport on winter road	The number of annual return truck trips is estimated to be 2,800.	The 2013 estimate is 1,407 return trips carrying freight and fuel.

Source: Project Description Changes provided by De Beers, February 11, 2013.
WMP = water management pond; WTP = water treatment plant; m = metre; m^3/d = cubic meters per day.

2.4.1.3 Volume of Treated Effluent Discharged

The treated effluent discharge from the water treatment plant (WTP) and the temporary water treatment plant (TWTP) at Snap Lake Mine in 2012[4] is summarized by month in Table 2-4. A total of 10.7 million cubic metres (Mm^3) of treated effluent was discharged in 2012, which is an increase of 27% over 2011. The 2012 daily treated effluent discharge, which includes the WTP and the TWTP, is shown in Figure 2-3.

[4] "2012 License Reporting Requirements" provided by Be Beers January 20, 2013.

De Beers increased the volume of treated effluent released to Snap Lake during the 2012 spring freshet. This was done to maintain on-site water levels at acceptable levels, and prevent spills to the environment (Golder 2012c). Modifications to the existing treated effluent discharge system increased the effluent discharge volume by approximately 10,000 cubic metres per day (m^3/d). A temporary floating diffuser was constructed and placed on the ice, directly above the permanent diffuser. The temporary floating diffuser provided an additional discharge capacity of approximately 8,000 m^3/d between May 20 and June 5, 2012. The maximum discharges occurred in the last week of May 2012 (Table 2-4; Figure 2-3).

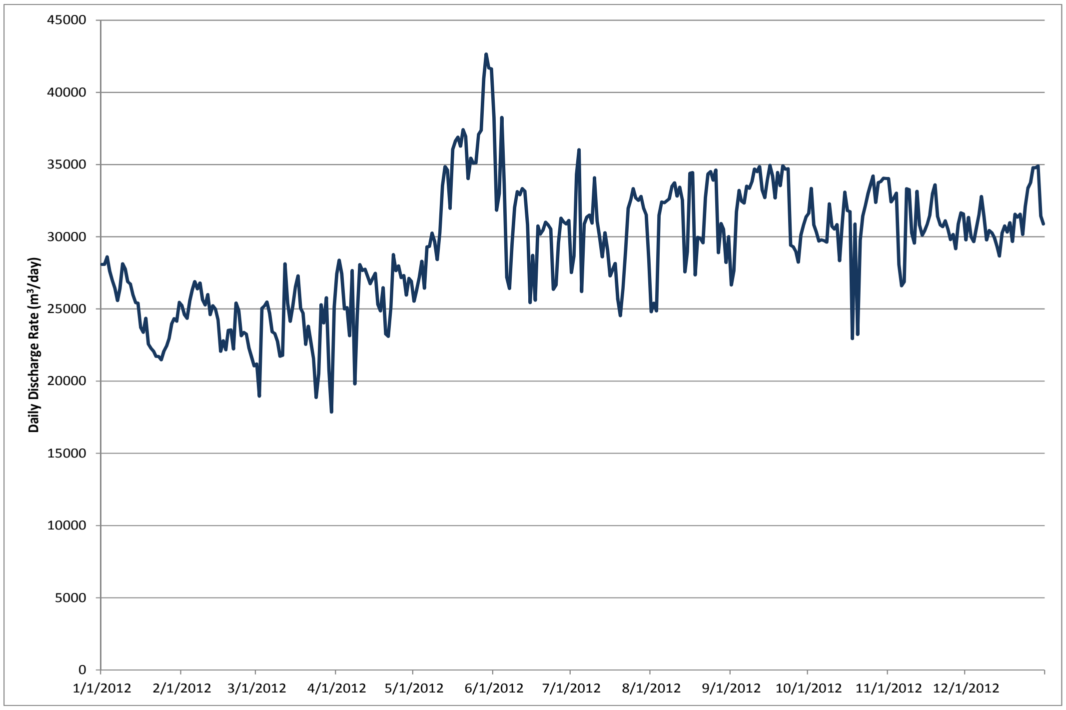
In 2012, the TWTP, which contributes to the volume of treated effluent discharged, only discharged water directly to Snap Lake in March.

Table 2-4 2012 Treated Effluent Discharge from the Snap Lake Mine

Month of Discharge (2012)	Average Discharge (m^3/d)	Maximum Discharge (m^3/d)	Minimum Discharge (m^3/d)	Total Discharge (m^3)
January	24,900	28,600	21,500	772,800
February	24,300	26,900	21,000	703,500
March	23,500	28,100	17,800	728,800
April	26,300	28,800	19,800	787,500
May	33,800	42,700	25,500	1,047,200
June	30,800	38,300	25,400	923,900
July	30,200	36,000	24,500	937,100
August	31,100	34,600	24,800	963,300
September	32,400	35,000	26,700	971,100
October	31,100	34,200	22,900	964,400
November	31,000	34,000	26,600	931,300
December	31,200	34,900	28,700	968,100
Total 2012	**29,200**	**42,700**	**17,800**	**10,699,000**

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

Figure 2-3 Daily Treated Effluent Discharge from the Snap Lake Mine, 2012



Note: m^3/day = cubic metres per day

2.4.2 Meteorological Monitoring

Wind conditions, relative humidity, temperature, precipitation, and solar radiation for the Snap Lake Hill and Lake meteorological stations in 2012 are included in Appendix 2A, Figures 2A-1 to 2A-14. Rainfall data were not collected at the Lake Station due to equipment malfunction.

Similar to previous years, predominant winds at the Mine were from the east and east-southeast in 2012. Lower wind speeds were measured from the northwest. The windroses depicting the wind speeds and directions from Hill and Lake Stations are presented in Appendix 2A, Figure 2A-1, Figure 2A-2, Figure 2A-8, and Figure 2A-9.

The total annual precipitation recorded at the Hill Station for Snap Lake in 2012 was 138.9 millimeters (mm), which is approximately 8.7% lower than the Yellowknife total for 2012 (150.8 mm) and 18.6% lower than the Yellowknife long-term (1971 to 2000) annual precipitation average of 164.5 mm (Environment Canada 2012). Rainfall followed the same pattern observed in the past five years. The 2012 precipitation trends for the Hill and Lake Stations are presented in Appendix 2A, Figure 2A-6, and Figure 2A-13.

The average annual temperature of -6.7 degrees Celsius (°C) in 2012 for Snap Lake Hill Station was 2.1°C colder than the annual temperature of -4.6°C for Yellowknife during 1971 to 2000. The

2012 monthly air temperature average in Yellowknife was warmer than the long-term average temperature except for April, November, and December (Environment Canada 2012). Yellowknife was 1.5°C warmer in 2012 (-3.1°C), than the long-term average of -4.6°C (1971 to 2000). The temperature data for the Lake and Hill Stations are presented in Appendix 2A, Figure 2A-5 and Figure 2A-12

The data for relative humidity for the Snap Lake Hill and Lake stations are consistent with the patterns and ranges of the Yellowknife data. 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 relative humidity data for the Snap Lake Hill and Lake stations are presented in Appendix 2A, Figure 2A-3, Figure 2A-4, Figure 2A-10, and Figure 2A-11.

2.4.3 Hydrological Monitoring

The surveyed water elevations and the range of minimum and maximum water surface elevations between 2002 and 2012 for Snap Lake and the reference lakes are provided in Table 2-5. Snap Lake had a lower range of elevation changes between 2002 and 2012, than 1999 Reference Lake, North Lake, and Northeast Lake, indicating that the Mine operations likely had a minimal effect on fluctuations in the Snap Lake water surface elevation.

The water surface elevation of Snap Lake increased between 2011 and 2012, and showed less variability over this time period than the reference lakes. At the last open-water survey during 2012, Snap Lake remained within the range of water surface elevations measured between 2002 and 2011. Water surface elevation data for the four lakes are shown in Table 2-5 and Figure 2-4.

Table 2-5 Surveyed Water Elevations for Snap Lake and Reference Lakes

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

Table 2-5 Surveyed Water Elevations for Snap Lake and Reference Lakes

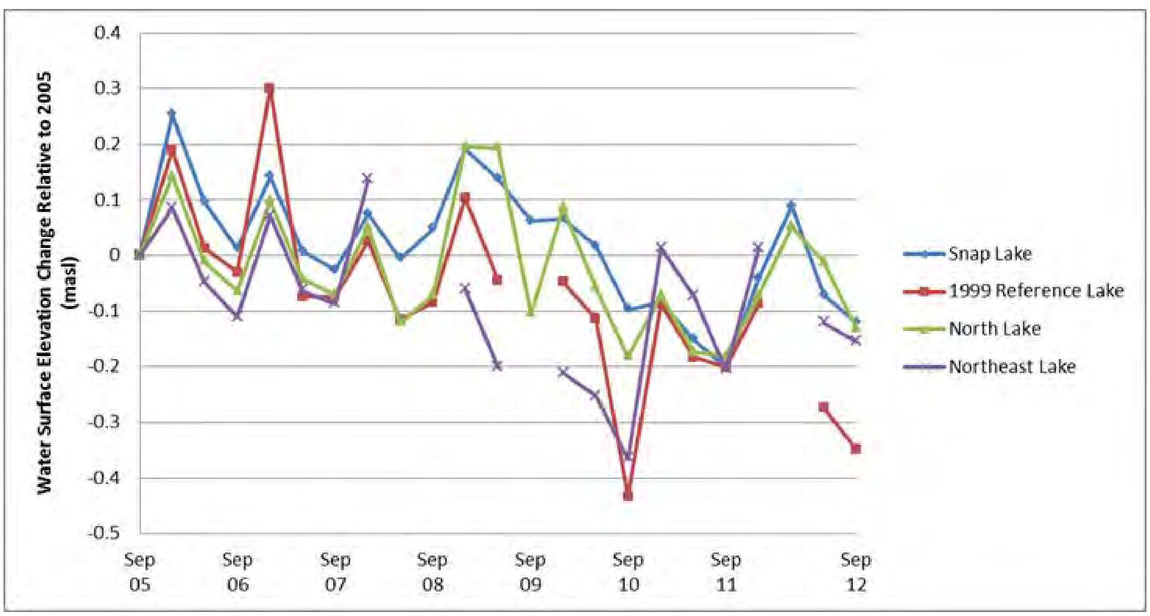
Year	Month	Snap Lake (masl)	1999 Reference Lake (masl)	North Lake (masl)	Northeast Lake (masl)
2010	June	444.217	440.729	439.852	432.760
2010	July	444.168	440.662	439.708	432.719
2010	September	444.054	440.343	439.584	432.607
2011	May	444.068	440.689	439.695	432.985
2011	July/August	444.000	440.593	439.592	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)]
Year-on-year change, 2011 to 2012	September 2011 to September 2012	+0.079	-0.148	+0.049	+0.050
Range between maximum and minimum surveyed water levels, 2002 to 2012	All months	0.453	0.734	0.378	0.501

(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.

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

Figure 2-4 Water Surface Elevations of Snap Lake, 1999 Reference Lake, North Lake, and Northeast Lake, 2005 to 2012

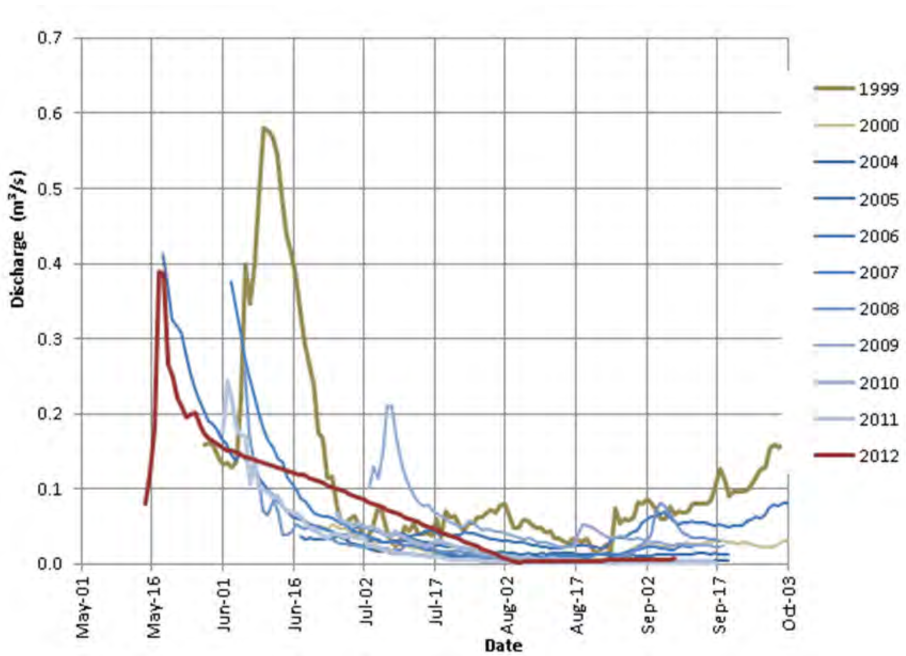


masl = metres above sea level.

2.4.3.1 Snap Lake

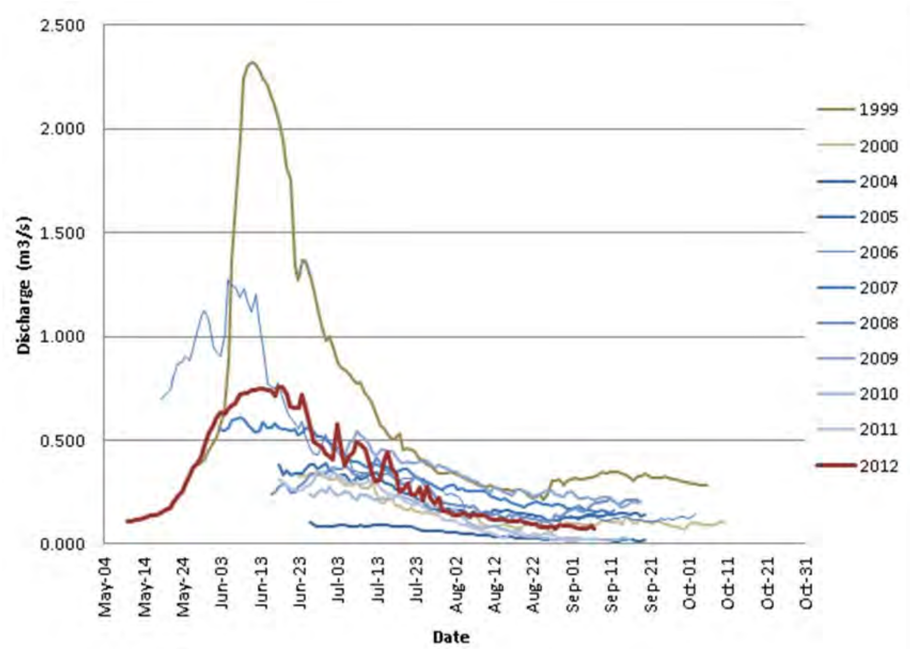
The discharges at the inflow (H4) station and outflow (H1 and H2) stations from 1999 to 2012 are shown in Figures 2-5 and 2-6. Inflows and outflows were within historical norms. Peak freshet during 2012 occurred on May 18 at the H4 (Stream 1) inflow station (Figure 2-5). Peak outflow at H1 and H2 occurred approximately between June 10 and June 24, 2012.

Figure 2-5 Discharge at Snap Lake Inflow (Station H4), 1999 to 2012



Note: m^3/s = cubic metres per second

Figure 2-6 Discharge at Snap Lake Outflow (Stations H1 and H2), 1999 to 2012



Note: m^3/s = cubic metres per second

Outflow discharge measurements for Snap Lake are provided in Table 2-6. The water elevation of Snap Lake increased by approximately 0.079 m between 2011 and 2012, and remained within the range of elevations surveyed between 2002 and 2011.

Table 2-6 Outflow Discharges for Snap Lake (Stations H1 and H2)

Date	Discharge [m^3/s]
29-May-2001	0.598
09-Jun-2002	0.415
12-Aug-2002	0.365
01-Oct-2002	0.250
26-Jun-2004	0.174
21-Sep-2004	0.043
18-Jun-2005	0.410
20-Sep-2005	0.145
19-May-2006	0.658
03-Aug-2006	0.279
03-Oct-2006	0.189
03-Jun-2007	0.516
15-Aug-2007	0.277
12-Sep-2007	0.202
09-Jun-2008	0.313
13-Aug-2008	0.115
18-Sep-2008	0.164
02-Jul-2009	0.481
24-Aug-2009	0.258
19-Sep-2009	0.220
23-Jun-2010	0.211
31-Jul-2010	0.182
16-Sep-2010	0.035
28-May-2011	0.142[a]
31-Jul-2011	0.128
18-Sep-2011	0.032
28-May-2012	0.348
3-Aug-2012	0.184
7-Sep-2012	0.087

(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 2-7. The water elevation of 1999 Reference Lake decreased by approximately 0.148 m between 2011 and 2012, and remained within the range of elevations surveyed between 2002 and 2011.

Table 2-7 Measured Water Elevation and Outflow Discharges for 1999 Reference Lake

Date	Geodetic Elevation (masl)	Discharge (m^3/s)
07-Jul-2002	440.839	0.423
11-Aug-2002	440.846	0.340
30-Sep-2002	440.839	0.311
27-Jun-2004	440.770	0.160
21-Sep-2004	440.652	0.060
18-Jun-2005	440.869	0.667
25-Aug-2005	440.699	0.086
19-Sep-2005	440.759	0.199
20-May-2006	440.966	1.443
03-Aug-2006	440.789	0.250
02-Oct-2006	440.746	0.138
02-Jun-2007	441.077	0.815
14-Aug-2007	440.703	0.191
12-Sep-2007	440.702	0.131
09-Jun-2008	440.803	0.691
13-Aug-2008	440.661	0.073
17-Sep-2008	440.692	0.103
02-Jul-2009	440.880	0.925
17-Aug-2009	440.732	0.178
09-Sep-2009	n/a	0.129
24-Jun-2010	440.729	0.193
31-Jul-2010	440.662	0.080
15-Sep-2010	440.343	0.012
28-May-2011	440.689	0.290
01-Aug-2011	440.593	0.033
18-Sep-2011	440.575	0.023
28-May-2012	440.689	0.302
5-Aug-2012	440.502	0.138
8-Sep-2012	440.427	0.055

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 2-8. Data from the 2012 water elevation surveys for North Lake were neither consistent with those from previous years nor with the corresponding discharge data. Therefore, the water elevations for North Lake were back-calculated using the stage-discharge rating curve. The water elevation of North Lake decreased by approximately 0.049 m between 2011 and 2012, and remained within the range of elevations surveyed between 2002 and 2011.

Table 2-8 Measured Water Elevation and Outflow Discharges for North Lake

Date	Geodetic Elevation (masl)	Discharge (m^3/s)
08-Jul-2002	439.865	0.087
11-Aug-2002	439.846	0.072
30-Sep-2002	439.807	0.046
25-Jun-2004	439.784	n/a
21-Sep-2004	439.652	0.012
17-Jun-2005	439.865	n/a
25-Aug-2005	439.727	n/a
19-Sep-2005	439.705	0.022
20-May-2006	439.909	0.128
03-Aug-2006	439.755	0.046
02-Oct-2006	439.702	0.025
03-Jun-2007	439.870	0.093
14-Aug-2007	439.723	0.026
12-Sep-2007	439.696	0.018
09-Jun-2008	439.817	n/a
13-Aug-2008	439.645	0.021
17-Sep-2008	439.695	0.020
01-Jul-2009	439.962	0.146
17-Aug-2009	439.960	0.078
18-Sep-2009	439.661	0.011
23-Jun-2010	439.852	0.055
31-Jul-2010	439.708	0.034
15-Sep-2010	439.584	0.005
28-May-2011	439.695	n/a
01-Aug-2011	439.592	0.007
18-Sep-2011	439.585	0.002
28-May-2012	439.695	n/a
6-Jul-2012	439.818[(a)]	0.052
5-Aug-2012	439.754[(a)]	0.027
8-Sep-2012	439.634[(a)]	0.008

(a) Elevations calculated using stage-discharge rating curve and measured discharge flows since survey data were incorrect.

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 2-9. Data from the September 2012 water level survey for Northeast Lake were neither consistent with those from previous years nor with the corresponding discharge data. Therefore, the September water surface elevation for Northeast Lake was back-calculated using the stage-discharge rating curve. The water elevation of Northeast Lake decreased by approximately 0.050 m between 2011 and 2012, and remained within the range of elevations surveyed between 2002 and 2011.

Table 2-9 Measured Water Elevations and Outflow Discharges for Northeast Lake

Date	Geodetic Elevation (masl)	Discharge (m^3/s)
08-Jul-2002	433.117	1.373
11-Aug-2002	433.068	0.754
30-Sep-2002	433.037	0.526
25-Jun-2004	432.993	0.107
21-Sep-2004	432.877	0.080
18-Jun-2005	433.102	1.592
25-Aug-2005	432.917	0.228
20-Sep-2005	432.897	0.227
20-May-2006	433.057	1.055
03-Aug-2006	432.924	0.251
03-Oct-2006	432.861	0.137
02-Jun-2007	433.043	0.653
14-Aug-2007	432.909	0.242
11-Sep-2007	432.885	0.160
08-Jun-2008	433.108	1.349
13-Aug-2008	n/a	0.187
17-Sep-2008	n/a	0.142
01-Jul-2009	432.911	1.582
17-Aug-2009	432.771	0.378
18-Sep-2009	n/a	0.243
23-Jun-2010	432.76	0.322
30-Jul-2010	432.719	0.119
15-Sep-2010	432.607	0.022
28-May-2011	432.985	0.238
31-Jul-2011	432.899	0.035
18-Sep-2011	432.767	0.041
28-May-2012	432.985	0.241
5-Aug-2012	432.851	0.232
8-Sep-2012	432.817[(a)]	0.057

(a) Elevations calculated using stage-discharge rating curve and measured discharge flows since survey data was incorrect.

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 2012 were within values recorded between 1999 and 2011 and are considered within normal ranges. Water elevations for Snap Lake, North Lake, and Northeast Lake increased between 0.049 m and 0.078 m between 2011 and 2012, 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 2012 were also considered to be within normal historical ranges. The water elevation of 1999 Reference Lake decreased between 2011 and 2012; it is not clear why this lake followed a different trend than the other lakes.

2.4.4 Water Temperature Monitoring

Water temperature data collected from the temperature loggers for Snap Lake, Northeast Lake, and Lake 13 are presented in Figures 2-7 to 2-9.

The shallow sample site temperature loggers (Figure 2-7) followed a similar trend in Snap Lake, Northeast Lake, and Lake 13 through the late spring and summer. The water temperature in Northeast Lake was lower than the other lakes during the first 10 days of measurement.

For the deep sample sites, the surface temperature loggers were recovered only for Northeast Lake and Lake 13. The surface temperatures at the deep sites in Northeast Lake and Lake 13 showed a similar pattern through the late spring and summer (Figure 2-8); however, Northeast Lake surface temperatures were consistently several degrees cooler than Lake 13.

The deep temperature logger at Northeast Lake (Figure 2-9) showed a seasonal trend similar to the shallow and surface locations. The deep temperature loggers did not produce a reliable data set for Snap Lake; they did not match temperatures measured by AEMP field programs.

Reference Lake 13 temperature measurements were consistently low (i.e., around 11°C) early in the season then sharply increased in mid-August to around 14.5°C (Figure 2-9). The mid-August temperatures were verified through comparison to temperatures measured by AEMP field programs. The cause of the sharp temperature increase in Lake 13 could potentially be wind-induced mixing or changes in the stratification of the lake.

Figure 2-7 Thermographs from Shallow Sample Sites (Total Depth <1 m)

Note ° C = degrees Celsius; m = metres

Figure 2-8 Thermographs from Deep Sample Sites - Surface Measurement (Sample Depth 0.3 m)

Note ° C = degrees Celsius; m = metres

Figure 2-9 Thermographs from Deep Sample Sites – Deep Measurement (Sample Depth 1 m above Bottom Substrate)

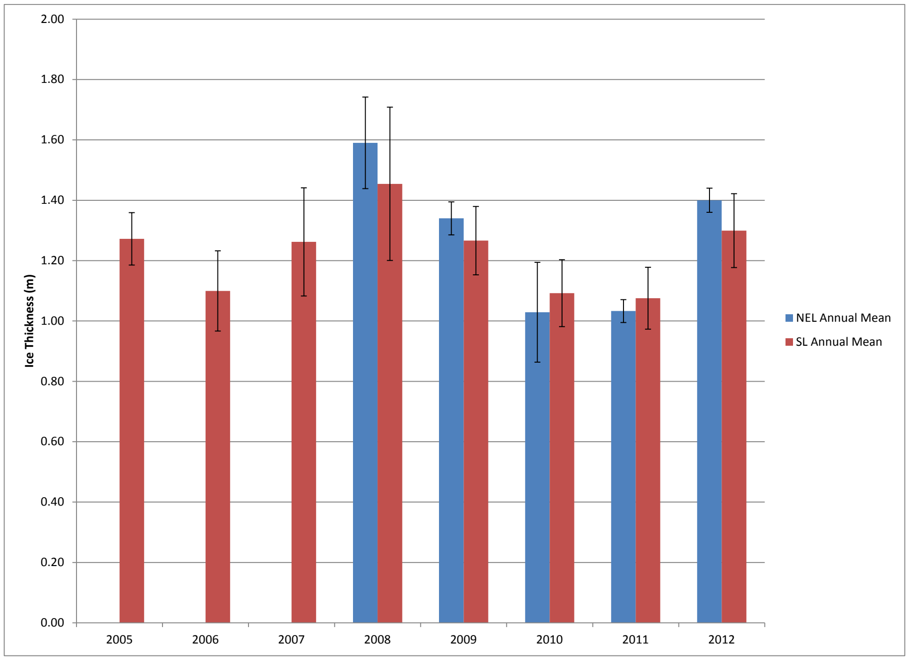
Note ° C = degrees Celsius; m = metres

2.4.5 Ice Thickness and Ice Cover Monitoring

2.4.5.1 Ice Thickness

Ice thickness data are available from 2005 for Snap Lake and from 2008 for Northeast Lake. Annual mean thicknesses for both lakes are presented by year in Figure 2-10. A Mann-Whitney U-test performed on the annual mean ice thickness (using data from 2008) found no statistically significant difference between ice thicknesses in the two lakes ($U = 12$, two tailed $p = 1.00$).

Figure 2-10 Ice Thickness-Annual Mean for Snap Lake and Northeast Lake



Note: Error bars indicate one standard deviation above and below the Annual Mean.

2.4.5.2 Days of Ice Cover Versus Open-water

The ice-off and ice-on dates for Snap Lake from 2008 to 2012 based on De Beers site staff field observations, are summarized in Table 2-10. The total days of ice cover in 2012 is 226, which is similar to the past four years (Table 2-10).

Table 2-10 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

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.

2.5 CONCLUSIONS

2.5.1 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?

From December 16, 2011 to December 15, 2012, 30 reportable spill events were recorded by De Beers site staff. Five spills with the potential to affect the aquatic environment were identified. The two largest volume spills (December 29, 2011 and August 1, 2012) were further described. The spill of 5,500 m^3 of process water from TS4 on December 29, 2011 was largely contained with physical mitigation measures. The release of 293 m^3 of treated effluent with elevated turbidity (i.e., greater than 7 NTU) on August 1, 2012 involved a short-duration event (i.e., less than 10 minutes) with a small volume of otherwise treated effluent. Both spills would have had negligible effects on the aquatic environment.

In 2012, 10.7 Mm^3 of treated effluent was discharged from the Mine. De Beers increased the volume of treated effluent released to Snap Lake during the 2012 spring freshet. The maximum discharges were in May, where treated effluent was discharged at rates up to 42,000 m^3/d during the last week of May 2012. The 2012 total treated effluent discharge was 27% higher than in 2011.

The Mine has undergone operational changes from the original project description. For example, freshwater is not used in the process plant. All surface water collection is directed to the WMP and not directly to the WTP. Water is not sprayed for dust suppression on the North Pile.

The total annual precipitation recorded at the Hill Station for Snap Lake in 2012 was 138.9 mm, which is approximately 8.7% lower than the Yellowknife total for 2012 (150.8 mm) and 18.6% lower than the Yellowknife long-term (1971 to 2000) annual precipitation average of 164.5 mm. Annual average temperatures were within the range of those observed in the past five years with the exception of April, November, and December, when the temperatures were lower than the long term climate minimum average.

The water surface elevation of Snap Lake increased between 2011 and 2012, but remained within the range measured from 2002 to 2011. The water surface elevation of Snap Lake varied less than the elevation of the three reference lakes, indicating that mine operations have a minimal impact on water surface elevation fluctuations. Peak freshet during 2012 occurred on May 18, with Snap Lake inflows and outflows that were within historic norms.

2.5.2 Is there a habitat difference between Snap Lake and the reference lakes in terms of seasonal water temperature and ice-cover?

2012 is the first year that water temperature loggers were installed at Snap Lake and the reference lakes. In July, the shallow site and deep site/surface temperature loggers indicated that Northeast Lake is several degrees cooler in temperature than Snap Lake and Lake 13. In the month of August, the temperature rises and falls in a similar pattern for Snap Lake and the reference lakes at shallow and surface sampling locations. For the deep measurement loggers, no data were available for Snap Lake, and different temperature patterns were observed in Northeast Lake and Lake 13.

There was no difference between Snap Lake and Northeast Lake in terms of mean annual ice thickness. Snap Lake had 226 days of ice cover in 2012 which is similar to the past five years.

2.6 RECOMMENDATIONS

The AEMP report should continue to review and consider spills and incidents which have the potential to affect the aquatic environment. Year-to-year changes to the project 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 to capture variations in spring temperatures. Redundancy should be built into the temperature logger program to verify data and reduce potential loss of data from field error and equipment malfunction.

Ice thickness measurements should be extended to Lake 13 and continued for Snap Lake and Northeast Lake.

De Beers site staff should continue to take descriptive and accurate notes related to ice cover on Snap Lake. Hydrological measurements should continue to be collected to record the peak of freshet.

2.7 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.

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Golder. 2012a. De Beers Snap Lake Mine Air Quality, Meteorological Monitoring and Emissions Reporting: 2012 Annual Report. Calgary, AB, Canada.

Golder. 2012b. De Beers Snap Lake Mine Streamflow and Lake Elevation Monitoring Program, 2012 Annual Report. Yellowknife, NWT, Canada.

Golder, 2012c. Snap Lake Spring Freshet 2012 Water Management and Mitigation - Summary Report. Calgary, AB, Canada.

3 WATER QUALITY

3.1 INTRODUCTION

3.1.1 Background

3.1.1.1 Snap Lake

Snap Lake is located about 200 kilometres (km) northeast of Yellowknife (Section 1, Figure 1-1). It is a relatively small lake, with a surface area of approximately 17 square kilometres (km^2) and a volume of 87 million cubic metres (Mm^3). The Snap Lake watershed (67 km^2) is located near the headwaters of the Lockhart River watershed (27,237 km^2), which drains into Great Slave Lake (Section 1, Figure 1-2).

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 from Mine dewatering activities began on June 22, 2004 using a temporary diffuser.

Between 2004 and June 14, 2012, De Beers conducted AEMP water quality monitoring to comply with requirements under Part F and G of the original Water Licence MV2001L2-0002 (MVLWB 2004), Section 5 of the Fisheries Authorization (Number SC00196-4.1; DFO 2004), and the Aquatic Effects Monitoring Plan (De Beers 2005a). A new Water License, MV2011L2-0004 (MVLWB 2012), was effective June 14, 2012, under which De Beers is currently conducting AEMP water quality monitoring.

3.1.1.2 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 License 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.

3.1.2 Objectives

The primary objectives of the water quality component of the AEMP are to:

- characterize and interpret water quality in Snap Lake in 2012;
- inform management decisions made by Mine personnel; and,
- verify and update the EAR predictions (De Beers 2002).

3.1.2.1 Key Questions

To meet the primary objectives of the AEMP water quality component, analyses and interpretation of water quality data 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[5], and Water Licence limits?
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?
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?

[5] 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).

The field survey and data analysis methods used to answer the key questions are described in Section 3.2. A summary of the quality assurance (QA) and quality control (QC) assessment on the 2012 data is provided in Section 3.3, followed by the 2012 results and conclusions (organized by key question) which are provided in Sections 3.4 and 3.5, respectively.

3.1.2.2 Special Studies

In addition to the core AEMP program, special studies occur as needed, 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. Special studies conducted during the 2012 AEMP with water quality components were:

- the Downstream Lakes Special Study (Section 12.2);
- the Reference Lake 13 Suitability Special Study (Section 12.3); and,
- the Nutrient Special Study (Section 12.4).

3.1.2.3 Other Water Quality Monitoring

In 2012, several targeted, short-term water quality monitoring programs were conducted to investigate whether specific Mine-related activities resulted in changes to water quality in Snap Lake. Such programs were conducted in response to:

- Spills 11-391/11-398 (Golder 2011, 2012a,b);
- a temporary increase in discharge of treated effluent to Snap Lake and use of a floating diffuser during spring freshet (Golder 2012c; De Beers 2012a);
- construction and operation of the IL6 diversion (Golder 2012c); and,
- replacement and operation of the permanent diffuser (Golder 2013).

Water quality data from the targeted programs listed above are referenced or discussed herein when findings are applicable to the AEMP data interpretation.

3.2 METHODS

3.2.1 Field Surveys for AEMP Sampling

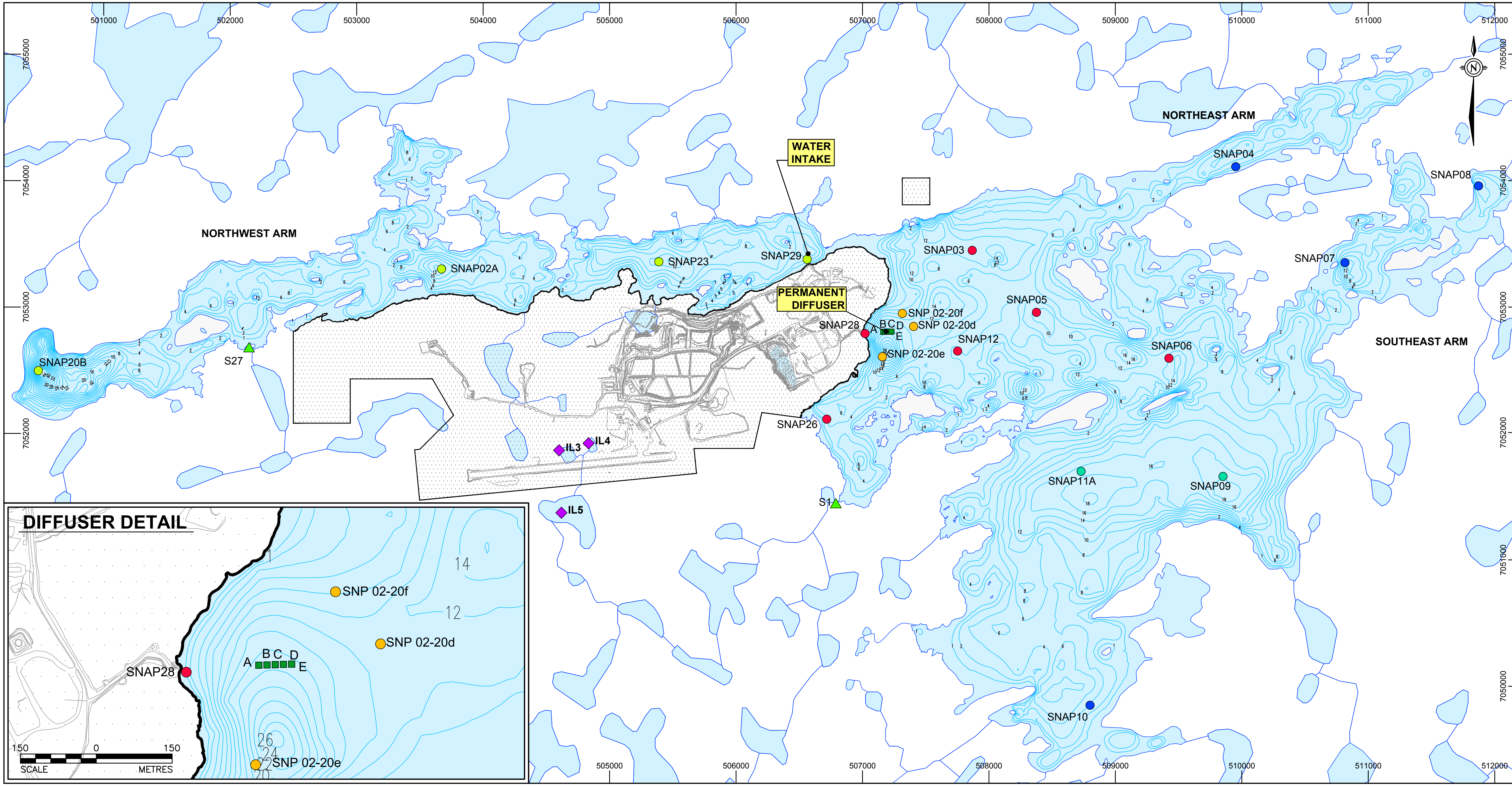
3.2.1.1 Locations of Sampling Stations

Thirty-one water quality stations were monitored during the 2012 AEMP water quality field programs (Figures 3-1, 3-2, and 3-3), excluding the special studies and targeted programs which are described in Section 3.2.1. Nineteen stations were within Snap Lake (Figure 3-1), and have been classified as diffuser, near-field, mid-field, far-field, and northwest arm stations. Stations were classified into these five different areas according to their geographical location relative to the diffuser outlet and historical influence of the minewater on water quality in Snap Lake. The diffuser, near-field, mid-field, and far-field stations are located in the largest basin, or "main basin" of Snap Lake. The northwest arm is connected to the main basin by a narrow area and has limited mixing with the main basin. The diffuser stations are the three deepest stations closest to the permanent diffuser (SNP 02-20d, SNP 02-20e, and SNP 02-20f), and are located within the diffuser's mixing zone (Figure 3-1).

The six near-field stations (SNAP03, SNAP05, SNAP06, SNAP12, SNAP26, and SNAP28) are located in the northern portion of the main basin of Snap Lake (Figure 3-1), where water quality has typically been most influenced by the discharge of treated effluent in this area (De Beers 2006, 2007a, 2008a, 2009, 2010, 2011a, 2012b), particularly during ice-covered conditions. Station SNAP28 is located near the diffuser outlet embankment, and SNAP05 and SNAP12 stations are located near the artificial reef, which is a fish habitat compensation structure. Near-field station SNAP14 was eliminated from the Snap Lake water quality monitoring program in April 2009 because data collected from other near-field stations were sufficient to characterize conditions in the near-field. Therefore, continued monitoring at SNAP14 was no longer required; however, SNAP14 continued to be monitored as part of the 2012 benthic invertebrate monitoring program (Section 5).

The two mid-field stations, SNAP09 and SNAP11A, are located farther away from the diffuser in the southern portion of the main basin (Figure 3-1). Prior to 2007, concentrations of treated effluent-related parameters were consistently lower at mid-field stations compared to the diffuser and near-field stations (De Beers 2007a, 2008a). In 2007, the treated effluent influence reached bottom waters of the mid-field area.

L:\2013\1349\13-1349-0001\Phase 5000\ Drawing file: 13134900015000A3-1.dwg Apr 23, 2013 - 12:02pm



LEGEND

- WATER QUALITY MONITORING STATION: DIFFUSER
- WATER QUALITY MONITORING STATION: NEAR-FIELD
- WATER QUALITY MONITORING STATION: MID-FIELD
- WATER QUALITY MONITORING STATION: FAR-FIELD
- WATER QUALITY MONITORING STATION: NORTHWEST ARM
- INLAND LAKE STATION
- DIFFUSER NOZZLE LOCATION
- TRIBUTARY TO SNAP LAKE STATION
- SNAP LAKE MINE FOOTPRINT
- WATERBODY
- DEPTH CONTOUR (m)

REFERENCE

DIGITAL MAP FROM MACKAY LAKE, NORTHWEST TERRITORIES, PRODUCED BY DEPARTMENT OF ENERGY, MINES AND RESOURCES. MAP 75M, ORIGINAL SCALE 1:250,000, NAD 83 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. ALL DEPTH VALUES ARE SHOWN IN METRES.
2. MAP MOVED TO NAD83 COORDINATE SYSTEM.

500 0 500
SCALE METRES

PROJECT

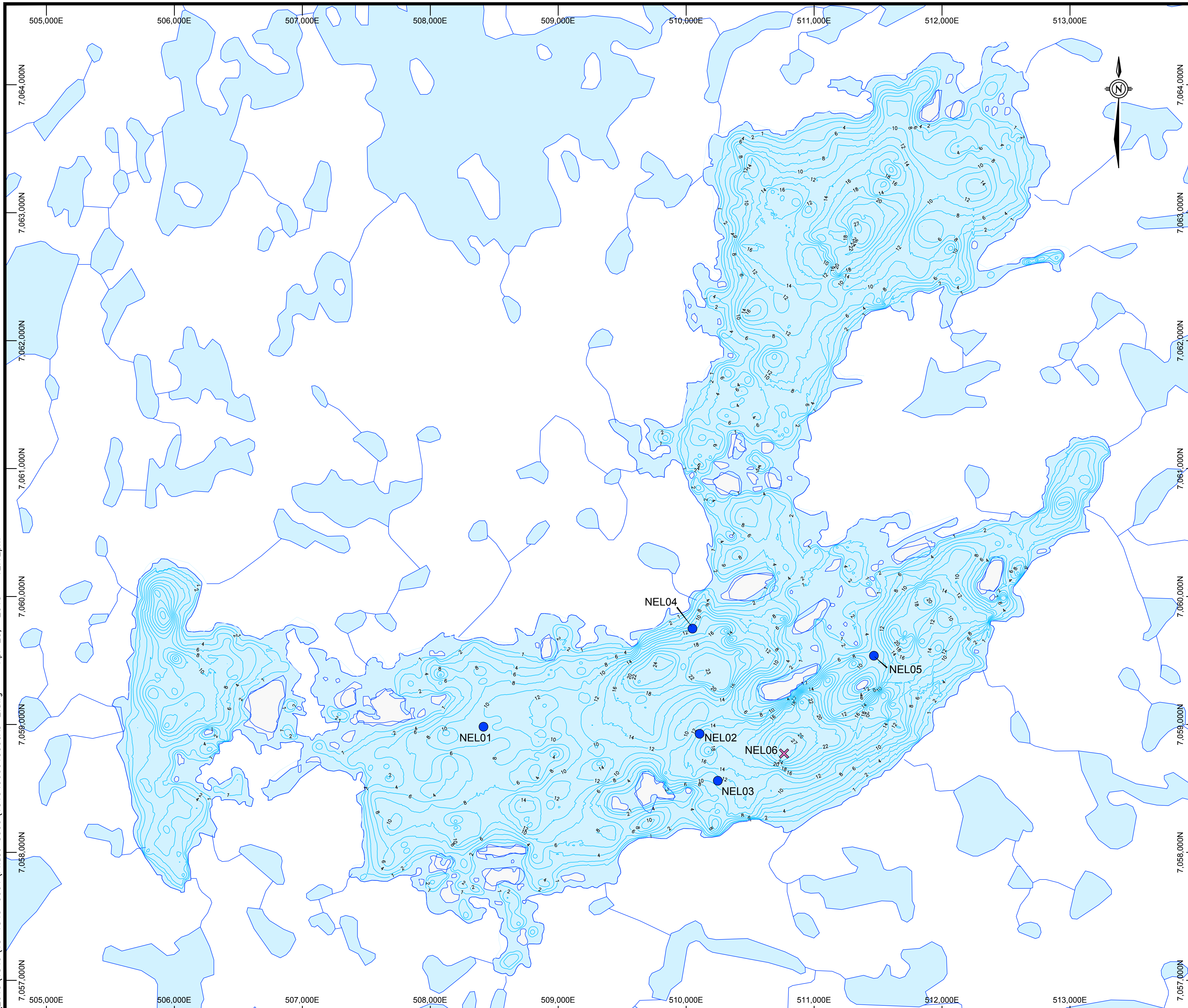
DE BEERS GROUP OF COMPANIES

TITLE

WATER QUALITY MONITORING STATIONS IN AND ADJACENT TO SNAP LAKE, 2012

Golder Associates

PROJECT	13-1349-0001.5000		FILE No. 13134900015000A3-1
DESIGN	TD	2013-03-13	SCALE AS SHOWN REV. 0
CADD	JEF	2013-03-29	
CHECK	LMB	2013-04-19	**FIGURE 3-1**
REVIEW	PMC	2013-04-19	



LEGEND

- WATER QUALITY AEMP MONITORING STATION
- WATER QUALITY PROFILE STATION
- DEPTH CONTOUR (m)
- WATERBODY

REFERENCES

DIGITIZED FROM NTS TOPOGRAPHIC MAP 75 M/10 © 1985 HER MAJESTY THE QUEEN IN RIGHT OF CANADA. DEPARTMENT OF ENERGY, MINES AND RESOURCES. PROJECTION : TRANSVERSE MERCATOR, DATUM : NAD27, COORDINATE SYSTEM : UTM ZONE 12.

REFERENCE 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.

500 0 500
SCALE METRES

PROJECT

DE BEERS
GROUP OF COMPANIES

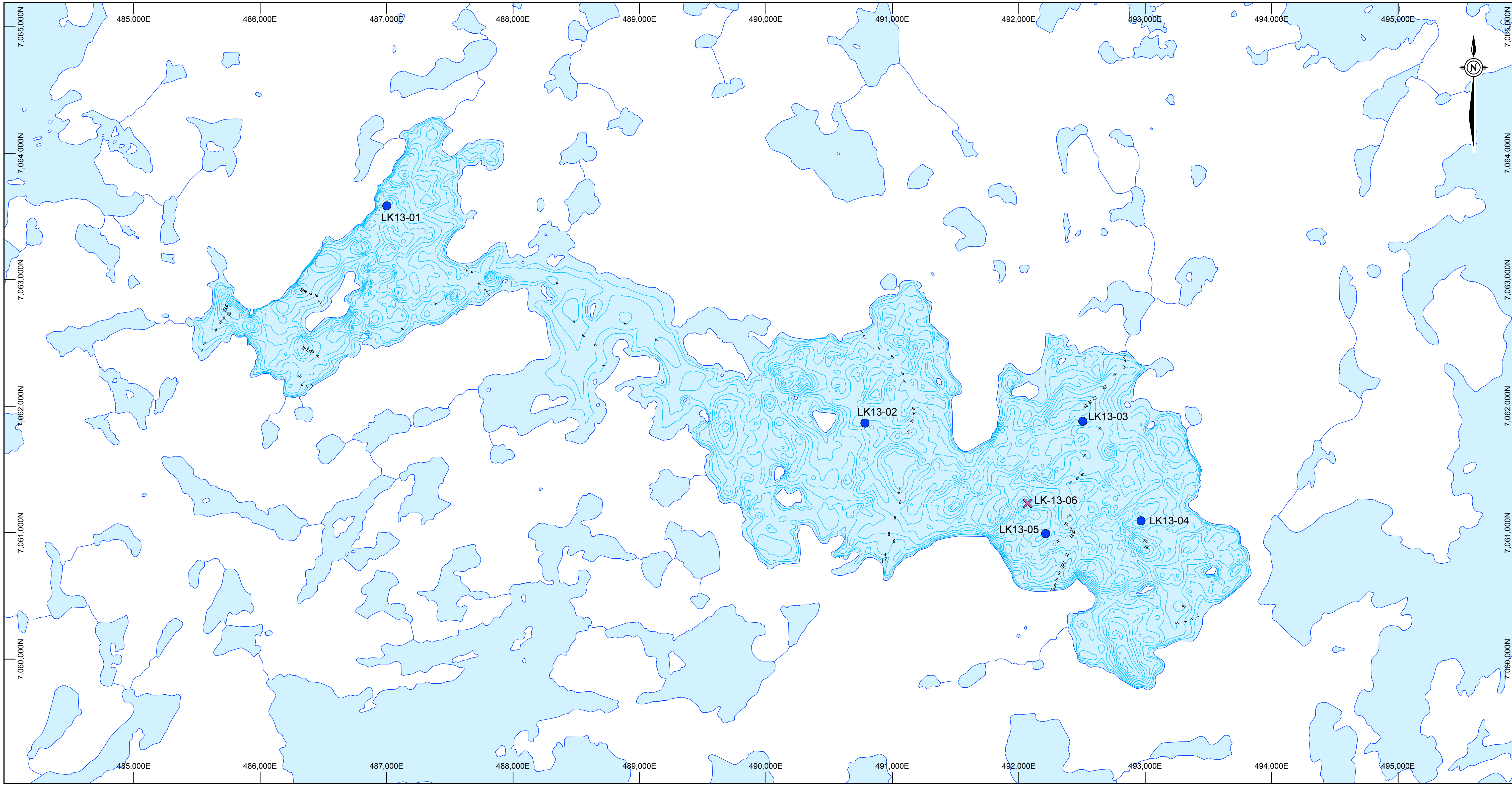
TITLE

WATER QUALITY MONITORING STATIONS IN NORTHEAST LAKE, 2012

Golder Associates

PROJECT	13-1349-0001.5000		FILE No.	13134900015000A3-2
DESIGN	TD	2013-03-13	SCALE AS SHOWN	REV. 0
CADD	JEF	2013-03-29		
CHECK	LMB	2013-04-19	FIGURE: 3-2	
REVIEW	PMC	2013-04-19		

L:\2013\1349\13-1349-0001\Phase 5000\13134900015000A3-2.dwg Apr 23, 2013 - 12:12pm



500 0 500
SCALE METRES

LEGEND

- WATER QUALITY MONITORING STATION
- WATER QUALITY PROFILE STATION
- WATERBODY
- DEPTH CONTOUR (m)

REFERENCE

BASE VECTOR DIGITAL CANVEC 1:50K DATA OBTAINED FROM GEOGRATIS ONLINE.

DATUM: NAD83 PROJECTION: UTM ZONE 12

PROJECT

DE BEERS
GROUP OF COMPANIES

TITLE

WATER QUALITY MONITORING STATIONS IN PROVISIONAL REFERENCE LAKE 13, 2012

Golder Associates

PROJECT	13-1349-0001.5000		FILE No.	13134900015000A3-3
DESIGN	TD	2013-03-13	SCALE AS SHOWN	REV. 0
CADD	JEF	2013-03-30		
CHECK	LMB	2013-04-19	**FIGURE: 3-3**	
REVIEW	PMC	2013-04-19		

L:\2013\1349\13-1349-0001\Phase 5000\ Drawing file: 13134900015000A3-3.dwg Apr 23, 2013 - 12:04pm

The four far-field stations are SNAP04, SNAP07, SNAP08, and SNAP10 (Figure 3-1). Three of the four far-field stations (SNAP04, SNAP07, and SNAP08) are located in relatively isolated long and narrow bays in the northeast portion of the main basin and one station, SNAP10, is located in the southernmost embayment of Snap Lake. Station SNAP08 is located at the Snap Lake outlet. The treated effluent influence reached bottom waters of the far-field area for the first time in the winter of 2009.

Four water quality stations (SNAP02A, SNAP20B, SNAP23, and SNAP29) are located in the northwest arm of Snap Lake (Figure 3-1). The water quality in the northwest arm has generally been the least influenced by treated effluent, likely because this 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 due to ice blockage over much of the narrows. However, water quality in the northwest arm has been increasingly influenced by treated effluent. Station SNAP29 is located near the water intake embankment and is closest to the main basin of Snap Lake. Sampling at SNAP23 began in April 2007 to increase the amount of water quality information from the northwest arm, in particular to monitor effects from potential seepage or overland flow from the wetlands that received treated domestic waste water from the water treatment system. Releases to the northwest arm from the domestic waste water treatment system have not occurred since 2009. However, monitoring in the northwest arm has continued to identify trends related to treated effluent exposure, potential seepage, and the untreated release from the waste rock pile collection sumps (i.e., Spills 11-391/11-398).

Water quality profile data were collected at all of the aforementioned water quality stations for the following field parameters: pH; specific conductivity, hereafter referred to as conductivity; dissolved oxygen (DO); and, water temperature. These field data were also collected at ten additional stations in Snap Lake as part of the plankton and benthic invertebrate monitoring programs (Sections 5 and 6):

- three near-field stations (SNAP13 - Section 5, Figure 5-1; and, SNAP14 and SNAP15 - Section 5, Figure 5-1);
- three mid-field stations (SNAP17, SNAP18, and SNAP19 - Section 6, Figure 6-1); and,
- four stations in the northwest arm of Snap Lake (SNAP01, SNAP30, and SNAP31 - Section 3, Figure 3-1; and SNAP20 - Section 6, Figure 6-1).

The field methods used for collecting field water quality profiles at the benthic invertebrate and plankton stations, including monitoring frequency, are discussed in more detail in their respective field survey sections (Sections 3 and 5).

Twelve stations located outside Snap Lake were also sampled as part of the core AEMP in 2012:

- Station KING01, located approximately 25 km downstream of Snap Lake in the Lockhart River system, upstream of King Lake (Figure 1-2). Monitoring at KING01 is conducted to evaluate water quality at a location downstream of Snap Lake.
- Three inland lake stations (IL3, IL4, and IL5), which 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, which 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 the reference lake, Northeast Lake (NEL01, NEL02, NEL03, NEL04, and NEL05), and one DO profile station, NEL06 (Figure 3-2). The water quality stations in Northeast Lake are monitored to identify local water quality changes that may not be influenced by Mine activities. The DO profile station in Northeast Lake is monitored to compare DO concentrations at similar depths in Snap Lake.

3.2.1.2 Water Quality Monitoring to Support Special Studies

3.2.1.3 Downstream Lakes Special Study

The 2010 AEMP report (De Beers 2011a) recommended that the focus of the AEMP be shifted from spatial and seasonal trends in Snap Lake to temporal changes and changes downstream of Snap Lake. In 2011, the initial downstream reconnaissance sampling program was completed in the first three lakes downstream of Snap Lake, Downstream Lake (DSL) 1, DSL 2, and Lac Capot Blanc. In 2011, signatures of treated effluent were evident in those three downstream lakes. Therefore, the Downstream Lakes Special Study was conducted in August 2012 to collect information (i.e., bathymetry, water quality, sediment, plankton, and benthic invertebrate) from the first three lakes downstream of Snap Lake and to further document the extent of treated effluent downstream of Snap Lake. One additional water quality sample was collected from DSL1 in July 2012 as part of the fish health program. The water quality information collected during the program was also used to support the process of selecting additional downstream stations for the draft AEMP Design Plan (De Beers 2012c). The methods and results of the 2012 Downstream Lakes Special Study are presented in Section 12.1.

3.2.1.4 Reference Lake 13 Suitability Special Study

Based on information collected during a review of potential reference lakes in 2005 (Golder 2005a,b), Provisional Reference Lake 13, herein referred to as Lake 13, was determined to be the second most similar lake to Snap Lake, following Northeast Lake, on the basis of size, shape, and physical characteristics. Inclusion of a second reference lake in the AEMP study design provides a better basis upon which to determine whether changes in Snap Lake are natural or Mine-related. The Reference Lake 13 Suitability Special Study collected baseline bathymetry, water quality, sediment, plankton, and benthic invertebrate information in August 2012. Water quality profiles and samples were collected from five stations in the lake, one station, LK13-01, in the west basin, and four stations, LK13-02, LK13-03, LK13-04, and LK13-05, in the east basin. One deep water station, LK13-06, was profiled for dissolved oxygen comparisons. One additional water sample was collected from Lake 13 in July 2012 as part of the fish health sampling program.

Data collected from Lake 13 were combined with data from the existing reference lake, Northeast Lake, and were used to characterize reference lake conditions relative to Snap Lake (Section 3.4). Lake 13 is currently a provisional reference lake, and has not been approved for inclusion into the core AEMP. However, Lake 13 data have been integrated into the water quality assessment. Therefore, Lake 13 field methods (e.g., sampling frequency and analytical parameters) are presented in Sections 3.2.1.6 to 3.2.1.10 along with the core AEMP program.

3.2.1.5 Nutrient Special Study

The 2012 Nutrient Special Study assessed discrepancies between the nutrient data collected for the AEMP water quality and plankton programs observed from 2008 to 2011. The 2012 Nutrient Special Study involved:

- spike samples of known nutrient concentrations were sent to three different laboratories to assess the accuracy of each laboratory in measuring known nutrient concentrations;
- nutrient results of split samples were compared between laboratories to identify any patterns in differences between laboratories; and,
- nutrient results collected at different depths, as per the methods followed for the water quality and plankton programs, were compared to identify any differences in nutrient concentrations related to depth.

Detailed methods and results of the 2012 Nutrient Special Study are presented in Section 12.3.

3.2.1.6 Sampling Frequency

This 2012 AEMP report includes water quality monitoring data collected between November 1, 2011 and October 31, 2012. The reporting period was chosen to allow complete ice-covered and open-water seasons to be analyzed together.

In 2012, the ice-covered season was defined as November 2011 to June 2012. The open-water season was defined as July to October 2012. The 2012 seasonal delineations are consistent with ice-covered and open-water seasons delineated in previous AEMP reports (De Beers 2006, 2007a, 2008a, 2009, 2010, 2011a, 2012b). In 2012, the January quarterly field program was re-scheduled to February due to extremely cold temperatures. Field crews sampled SNAP02A, SNAP03, SNAP20B, SNAP23, SNAP29, and KING01 stations in January 2012 before colder weather rendered sampling unsafe. The stations sampled in January 2012, with the exception of KING01, were sampled again in February 2012.

Since January 2007, surveys in June, October, November, and December have not been conducted because ice conditions are often unsafe. This modification to the initial AEMP sampling design followed consultation with the MVLWB (De Beers 2007b); however, the modification included contingencies so that if ice conditions were safe, sampling would be conducted. This was the case in 2009, when unseasonably good ice conditions allowed for completion of a June sampling program.

The frequency of sampling for each program area is outlined in Table 3-1. In 2012, stations were sampled at frequencies consistent with previous years. Stations were typically monitored on a quarterly, monthly, or weekly basis, as described below:

- **Quarterly:** The AEMP requires that all stations within Snap Lake and Northeast Lake, and the station downstream of Snap Lake, be monitored quarterly for all field and selected laboratory parameters (De Beers 2005a). Quarterly monitoring was completed twice during ice-covered conditions and twice during open-water conditions (Table 3-1).
- **Monthly:** The monthly collection of water quality samples and field water quality profiles at stations near the diffuser outfall is a requirement of both the AEMP (De Beers 2005a) and the Surveillance Network Program (SNP) in the Water Licence (MVLWB 2004, 2012). The 2004 Fisheries Authorization required monthly monitoring of field water quality profiles at all stations in Snap Lake from February to May (DFO 2004). These requirements have since been revised and incorporated in the 2012 Water Licence (MV2011L2-0004); water quality profiles are required at least two times per year during open-water and four times per year during ice-cover (MVLWB 2012). Monthly monitoring during open-water conditions of the three inland lakes (Table 3-1) was recommended in the EAR (De Beers 2002).
- **Weekly:** As outlined in De Beers (2005a) tributaries to Snap Lake are to be monitored at least weekly during spring freshet, and monthly during the open-water season (Table 3-1).

Additional details on requirements for sampling frequency for specific parameters are provided in Section 3.2.1.10.

Table 3-1 AEMP Water Quality Monitoring Frequency, 2012

Area	Sampling Stations	Frequency	2012 Sampling Period
Diffuser	SNP 02-20d, SNP 02-20e, and SNP 02-20f	Monthly	February 19
			March 18 and 21
			April 15 and 17
			May 13
			July 8
			August 12
			September 9
Near-field	SNAP03, SNAP05, SNAP06, SNAP12, SNAP26, and SNAP28	quarterly sampling (dates shown in **bold**) and monthly field water quality profiles during ice-covered conditions	**January 13[b] and 14[b]**
			February 17, 18, and 20
			March 16 and 17
			April 13, 16, and 17
Mid-field	SNAP09, SNAP11A		May 6 and 7
Far-field	SNAP04, SNAP07, SNAP08[a], SNAP10		**July 6, 7, 9, 11, 12, and 13**
Northwest arm	SNAP20B, SNAP02A, SNAP23, and SNAP29		**September 7, 11, 12, and 13**
Inland lake	IL3, IL4, and IL5	monthly during open-water conditions	July 14
			August 28
			September 15
Watercourse (major tributaries to Snap Lake)	S1, S27[c]	approximately twice weekly sampling and field measurements during spring freshet	May 14, 15, 18, 22, 25, and 28
		approximately monthly sampling and field measurements during open-water conditions	July 11
			September 1 and 7
Downstream[d]	KING01	quarterly sampling and field measurements	January 14[b]
			April 17
			July 13
			September 10
Northeast Lake	NEL01, NEL02, NEL03, NEL04, NEL05, and NEL06[e]	quarterly sampling (dates shown in **bold**) and monthly field water quality profiles during open-water conditions	**February 21**
			April 14
			July 10
			August 15
			September 8
Lake 13[f]	LK13-01, LK13-02, LK13-03, LK13-04, and LK13-05 and LK13-06[e]	sampled twice in 2012	July 10
			August 18, 19, 20, and 21

(a) SNAP08 is located at the Snap Lake outlet.

(b) Only SNAP02A, SNAP03, SNAP20B, SNAP23, SNAP29, and KING01 were sampled in January, as the program was rescheduled to February due to extremely cold temperatures that made further sampling unsafe.

(c) Monitoring Stream S27 was recommended in the 2013 draft AEMP Design Plan (De Beers 2012c). Stream S27 was sampled during spring freshet; no samples were collected in July or September 2012 (open-water).

(d) Additional downstream sampling was completed as part of the Downstream Lakes Special Study (Section 12.2)

(e) The Northeast Lake station NEL06 and Lake 13 station LK13-06 were added for deep water dissolved oxygen comparison.

(f) Lake 13 is a provisional reference lake and not currently formally part of the core AEMP.

3.2.1.7 Field Program Logistics

The Snap Lake and Northeast Lake 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 throughout the year. The inland lakes and S1 stations were accessed by truck and on foot during open-water conditions. Stream S27 was 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.

3.2.1.8 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[6] samples and a polyvinyl chloride (PVC) Kemmerer sampler for all other samples.

Diffuser Stations

Three samples were collected at each of the diffuser stations (SNP 02-20d, SNP 02-20e, and SNP 02-20f):

- one near the surface of the water, at approximately 0.3 metres (m) below the surface during open-water sampling or 0.3 m below the bottom of the ice layer during ice-covered sampling;
- one at the depth of maximum conductivity, or at mid-depth in the water column if a vertical conductivity gradient was not observed; and,
- one at 1.0 m above the lake bottom.

Other Snap Lake and Reference Lake Stations

At the northwest arm, near-field, mid-field, far-field, Northeast Lake, and Lake 13 stations, a check for a vertical conductivity gradient was completed at each station to determine the number of water samples to be collected (prior to water sample collection). A vertical conductivity gradient in the water column was identified by either of the following criteria:

[6] The term "metals" includes metalloids (e.g., arsenic) and non-metals (e.g., selenium).

- conductivity measurements throughout the water column at the station were less than 60 microSiemens per centimetre (µS/cm) and the difference in the conductivity range in the water column was greater than 15 µS/cm; or,
- maximum conductivity was greater than 60 µS/cm and the difference in the conductivity range was greater than 25 percent (%) of the midpoint.

Vertical conductivity gradients were not identified in Snap Lake, Northeast Lake, or Lake 13 in 2012; therefore, one mid-depth water sample was collected at each station.

Watercourse, Inland Lake, and 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. Grab samples were also collected quarterly at the AEMP downstream station KING01. Surface-water grab samples were collected at approximately 0.3 m below the surface.

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. The water that required filtering was poured from the Kemmerer sampler into clean 1 litre (L) laboratory-grade sampling containers and filtered when the 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 3.2.1.10). 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.

Samples for analysis of ultra-low level cadmium were collected in individual 125 mL plastic bottles and shipped to Alberta Innovates Technology Futures (Alberta Innovates) for analysis (Section 3.2.1.10).

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 the 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 water treatment plant (WTP) between November 2011 and October 2012 for quarterly toxicity testing in accordance with Water Licence (MVLWB 2004, 2012) (i.e., January, April, June, and October). One additional treated effluent sample was collected from the WTP in September for comparison to toxicity results from lake samples collected at the diffuser stations on the same day. One treated effluent sample was also collected from the temporary water treatment plant (TWTP) in March 2012. All six treated effluent samples were submitted to HydroQual Laboratories (HydroQual) in Calgary, Alberta (AB) and tested for acute and chronic toxicity (Section 3.2.1.10).

In 2012, 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), sampling occurred once during ice-covered conditions (April) 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 3E.

3.2.1.9 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 Aquatic Effects Monitoring Program Station

Category	Station	Parameter
0.5 to 1 m depth profile intervals depending on the station depth, to 0.5 m or 1 m above the lake bottom.	lake stations (Snap Lake, Northeast Lake and Lake 13[a])	water temperature, DO, pH, conductivity
Single (spot) measurements	lake stations (Snap Lake, Northeast Lake and Lake 13[a])	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[b], and Inland Lake stations (IL3, IL4, IL5)	water temperature, DO, pH, conductivity, wind and weather conditions

(a) Lake 13 is a provisional reference lake and not currently formally part of the core AEMP.

(b) Monitoring Stream S27 was recommended in the 2013 draft AEMP Design Plan.

m = metre; DO = dissolved oxygen.

3.2.1.10 Laboratory Analyses

The water quality parameters, applicable sampling stations, and monitoring frequency of different parameter groups are summarized in Table 3-3.

The majority of water samples were submitted to ALS Canada Ltd. (ALS) in Edmonton, AB. Ultra-low level cadmium samples were submitted to Alberta Innovates in Vegreville, AB, and samples for ultra-low level mercury and methyl mercury analyses were submitted to Flett in Winnipeg, Manitoba (MB). Alberta Innovates and Flett were selected for the ultra-low level metals 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 Laboratories (HydroQual) in Calgary, AB. The parameter groups are defined in Table 3-3 and the analytical services provided by each laboratory in 2012 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 USEPA (2002) and USEPA (2001), respectively;
- Alberta Innovates in Vegreville: ultra-low level cadmium by microwave digestion and inductively coupled plasma mass spectrometry (ICP-MS);
- Taiga Environmental Laboratory in Yellowknife: *Escherichia coli* (*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 3E).

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 Water Quality Parameters, Stations, and Sampling Frequency

Parameter Categories	Parameter	Diffuser Stations	Snap Lake, KING01, and Northeast Lake Stations	Total Dissolved Solids (TDS) Stations	Lake 13	Inland Lake Stations	Watercourse Stations
		SNP 02-20d, SNP 02-20e, SNP 02-20f	SNAP02A, 03, 05, 06, 07, 08, 09, 11A, 20B, 23, 26, KING01, and NEL01, 02, 03, 04, 05	SNAP04, 10, 12, 28, 29	LK13-01, 02, 03, 04, 05	IL3, IL4, IL5	S1, S27[(a)]
AEMP and Water Licence							
Physical and conventional parameters	TSS, pH, turbidity, conductivity	monthly	quarterly	quarterly	twice[(b)]	monthly during open-water conditions	weekly during spring freshet and monthly after spring freshet during open-water conditions
TDS and major ions	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	quarterly	quarterly	twice	monthly during open-water conditions	twice weekly during spring freshet and monthly after spring melt during open-water conditions
Nutrients (standard)	TP and dissolved phosphorus, TOC, ortho-phosphate as P, total ammonia (as nitrogen [N]), nitrate (as N), nitrite (as N), nitrate/nitrite (as N), TKN (as N)	monthly	quarterly	quarterly for nitrate[(c)]	twice	monthly during open-water conditions for total ammonia (as nitrogen [N]); nitrate (as N); nitrite (as N); nitrate/nitrite (as N); TKN (as N)	weekly during spring melt and monthly after spring freshet during open-water conditions for total ammonia (as nitrogen [N]); nitrate (as N); nitrite (as N); nitrate/nitrite (as N); TKN (as N)
Nutrients (additional)	total and dissolved organic phosphorus, total and dissolved inorganic phosphorus	monthly	quarterly	not applicable	twice	not applicable	monthly during open-water conditions

Table 3-3 Summary of Water Quality Parameters, Stations, and Sampling Frequency

Parameter Categories	Parameter	Diffuser Stations	Snap Lake, KING01, and Northeast Lake Stations	Total Dissolved Solids (TDS) Stations	Lake 13	Inland Lake Stations	Watercourse Stations
		SNP 02-20d, SNP 02-20e, SNP 02-20f	SNAP02A, 03, 05, 06, 07, 08, 09, 11A, 20B, 23, 26, KING01, and NEL01, 02, 03, 04, 05	SNAP04, 10, 12, 28, 29	LK13-01, 02, 03, 04, 05	IL3, IL4, IL5	S1, S27[(a)]
Metals	total and dissolved metals (Al, Sb, As, Ba, Be, Bi, B, Cd, Cs, Cr, $Cr^{(VI+)}$ (total only), Co, Cu, Fe, Pb, Li, Mn, Hg, Mo, Ni, Se, Ag, Sr, Tl, Ti, U, V, Zn)	monthly	total metals were analyzed; dissolved metals samples were archived and only analyzed if a total metal was above a guideline	not applicable	total metals were analyzed; dissolved metals samples were archived and only analyzed if a total metal was above a guideline	not applicable	weekly during spring melt and monthly after spring freshet during open-water conditions
Other parameters	methyl mercury and BOD	monthly	not applicable, except BOD at SNAP08	not applicable	not applicable	not applicable	not applicable
Water Licence Only							
Organics	BTEX, total oil and grease, TEH, TVH, F1 (without BTEX) and F2 (without BTEX	monthly	not applicable	not applicable	not applicable	not applicable	not applicable
Microbiological	*Escherichia Coli (E. coli)*	monthly	not applicable	not applicable	not applicable	not applicable	not applicable
Chronic toxicity	*Ceriodaphnia dubia, Pseudokirchneriella subcapitata*	twice a year	not applicable	not applicable	not applicable	not applicable	not applicable

(a) S27 was not sampled during open-water conditions.

(b) Lake 13 was sampled in July and August 2012.

(c) Nitrate is required for the calculation of calculated total dissolved solids.

AEMP = Aquatic Effects Monitoring Program; TDS = total dissolved solids; TSS = total suspended solids; $SiO_{2;}$ = silicate; P = phosphorus; N = nitrogen; TP = total phosphorus; TOC = total organic carbon; TKN = total Kjeldahl nitrogen; BOD = biochemical oxygen demand; TEH = total extractable hydrocarbons; TVH = total volatile hydrocarbons; NEL = Northeast Lake; 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; $Cr^{(VI+)}$ = hexavalent chromium (total only); 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 2012 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 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 key questions are outlined in Table 3-4 and described in Section 3.2.2. The results and conclusions from the 2012 AEMP, organized by key question, are provided in Section 3.4 and Section 3.5, respectively.

Table 3-4 Overview of Analysis Approach for Water Quality Effects 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) are discussed, where appropriate.
2. Are concentrations of key water quality parameters in Snap Lake below AEMP benchmarks and Water Licence limits?	Average and maximum concentrations of water quality parameters were compared to AEMP benchmarks and Water Licence limits (e.g., TDS). Instances where concentrations were above AEMP benchmarks or limits were identified and qualitatively assessed for potential Mine-related causes.
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?	An analysis of temporal patterns in water quality was completed for DO, TP, and parameters that are significantly correlated with conductivity in Snap Lake. A statistical test (e.g., Seasonal Kendall or other appropriate test) was used where appropriate to quantify the certainty of any potential temporal trends identified from laboratory parameters. Comparisons were made to the normal range observed prior to treated effluent discharge as well as reference lake concentrations. The potential to exceed AEMP benchmarks, EAR predictions, or updated model results was assessed for parameters with apparent increasing trends (or decreasing trends as for dissolved oxygen) in Snap Lake.

Table 3-4 Overview of Analysis Approach for Water Quality Effects 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 exist, 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 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 estimate loadings to Snap Lake from the deposition of air emissions from the Mine.
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; AEMP = Aquatic Effects Monitoring Program; SNP = Surveillance Network Program.

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?

Treated Effluent

For treated effluent, temporal plots of discharge volume, parameter concentrations, and loadings (from both the WTP and the 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. These evaluations are discussed in more detail below.

Comparisons to Water Licence Limits

Parameters with Water Licence limits are total suspended solids (TSS), nitrogen compounds (ammonia, nitrate, and nitrite), ions (chloride, and sulphate), metals (aluminum, chromium, copper, lead, nickel, and zinc), and a metalloid (arsenic) (MVLWB 2012). For these parameters, the Water Licence specifies both a "maximum concentration in any grab sample" and an "average monthly limit". An average monthly limit is the concentration that cannot be exceeded, determined by averaging the analytical results of six consecutive samples collected at 6-day intervals over a 30-day period. For parameters measured every six days (i.e., physical parameters, major ions, nutrients) a 30-day moving average was calculated for comparison. For metals, which are analyzed approximately once per month, a monthly value was used.

The following additional limits apply at 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 fraction F1 (C_6-C_{10}) and 2.1 mg/L for fraction F2 (C_{11}-C_{16}); and,
- the total phosphorus (TP) annual load limit is 256 kilograms per year (kg/y).

Treated effluent data were plotted 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. The TP annual load to Snap Lake was calculated using the WTP and TWTP treated wastewater discharge. Phosphorus concentration data and coincident flow rate data from the WTP were used to calculate flow-weighted average concentrations. The average was then multiplied by the total volume of WTP and TWTP discharge, from November 2011 through to October 2012, to estimate the TP loading during that year. The TP loading to Snap Lake was then compared to the Water Licence limit of 256 kg/y. Similar to previous AEMP reports, the total loading to Snap Lake for phosphorus was calculated using Equation 3-1:

$$\text{Total TP Load}_{WTP} = (FWC_{WTP} \times V_{WTP}) + (FWC_{TWTP} \times V_{TWTP}) \qquad \text{[Equation 3-1]}$$

where:

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

V_{WTP} = total volume of discharge at SNP 02-17B (November 2011 to October 2012);

FWC_{TWTP} = flow-weighted average TP concentration in the treated effluent from the TWTP (SNP 02-17); and,

V_{TWTP} = total volume of discharge at SNP 02-17 (November 2011 to October 2012, if applicable).

Comparisons to Environmental Assessment Report Predictions

A summary of parameters for which flow-weighted concentrations exceeded EAR predictions was developed. Flow-weighted concentrations have been presented to provide values more reflective of average conditions, rather than instantaneous concentrations. Loadings were calculated for parameters with mass-based units; parameters such as pH and conductivity were excluded. The combined weighted average used for comparison to EAR predictions was calculated using Equation 3-2:

$FWC_{WTP} = \Sigma(C_{WTPi} \times F_{WTPi}) + \Sigma(C_{TWTPi} \times F_{TWTPi}) / \Sigma(F_{WTPi} + F_{WTPi})$ [Equation 3-2]

where:

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

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;

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

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

i = sampling event.

Biological data for the treated effluent samples, including bacterial counts of *E. coli* and fecal coliforms, are presented as geometric means. Bacteria reproduce at an exponential rate in domestic waste water. It is, therefore, 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 3-3:

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

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 2012 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 100% (v/v) sample, depending on the endpoint.

Dilution Factors

The permanent diffuser is intended to maximize the potential for initial mixing of the treated effluent discharged to Snap Lake. The diffuser does not influence total loadings to Snap Lake or lake-wide changes in water quality, but it can reduce TDS concentrations and concentrations of other constituents of the WTP discharge, near the diffuser. The estimated dilution factors achieved by the permanent diffuser were calculated using TDS concentrations in the WTP and TWTP discharge and parameter concentrations from the annual monitoring program in Snap Lake. Minimum dilution factors for the diffuser were calculated quarterly (i.e., February, May, July, and September), using Equation 3-4:

$$DF = (C_e - C_b)/(C_d - C_b) \qquad \text{[Equation 3-4]}$$

where:

DF = minimum dilution factor of the permanent diffuser;

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

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 near-field stations 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. Quality and quantity of uncontrolled runoff and groundwater are discussed in the 2012 Acid/Alkaline Rock Drainage (ARD) Appendix of the Annual Water Licence Report submitted to MVLWB in accordance with Water Licence (De Beers 2013a).

In 2011, an untreated release from the waste rock pile collection sumps to the northwest arm of Snap Lake (i.e., Spills 11-391 and 11-398) was noted in the 2011 AEMP annual report (De Beers 2012a). The spill response monitoring program began on October 2, 2011, immediately after the occurrence of the spill, and continued until June 18, 2012. The detailed methods and results of the spill response monitoring program were provided in Golder (2011, 2012a,b).

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

Water quality parameters 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 2012 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. Any site-specific benchmarks developed for Snap Lake (e.g., TDS, strontium, nitrate) as part of the AEMP Response Framework will be highlighted as such.

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 average conditions to WQGs, benchmarks, and predicted concentrations, or consideration of the information on which the aquatic life WQGs was developed.

Whole-Lake Average Licence Limit for 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 2012). In 2012, because all TDS concentrations were less than 350 mg/L, a simple mean of the depth-averaged means at all stations was used to calculate the whole-lake average. If, in future years, the depth-averaged concentration at any one 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.

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 2005, Method 2540). Alternatively, TDS concentrations can be calculated from the summation of major ions in the sample (APHA 2005). Calculated TDS was used rather than measured TDS for the Snap Lake AEMP consistent with recommendations in the Water Licence (MVLWB 2012) and for reasons outlined in Appendix 3A. TDS concentrations were calculated using Method 1030E

(APHA 2005). Further information on the TDS formula and its application in the AEMP water quality assessment is provided in Appendix 3A.

Toxicity Data

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 3E.

3.2.2.4 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?

Different methods were used to answer Key Question 3:

- screening for parameters that were positively correlated with conductivity and then visually evaluating temporal plots for these parameters at selected stations to identify increasing trends;
- using a statistical test to identify increasing trends for selected parameters at selected stations;
- comparing observed temporal trends with model predictions for key parameters;
- comparing maximum concentrations in Snap Lake with EAR predictions; and,
- reviewing vertical profiles of DO concentrations from different areas in Snap Lake over time.

Screening and 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, 2011a, 2012b).

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 2012. 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 2012, directly related to the input of treated effluent; and,
- conductivity was used to evaluate the degree of treated effluent exposure for other monitoring, 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 (148) 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. All parameters that significantly correlated with conductivity based on the inclusion or exclusion of the outliers were reviewed for temporal trends in Snap Lake.

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

- SNAP13 and SNP 02-20e (located near the permanent diffuser, at the edge of the mixing zone);
- SNAP05 (located in the near-field area);
- SNAP09 (located in the mid-field area);
- SNAP08 (located in the far-field area, near the outlet of Snap Lake);
- SNAP02 and SNAP02A (located in the northwest arm of Snap Lake); and,
- 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 SNAP 02A, respectively. Data from both the historical and new stations were included to provide a longer dataset for the analysis.

Temporal plots of TP, nitrate, nitrite, and ammonia at each of the above stations were also reviewed, regardless of the strength of correlations with conductivity. Because of seasonal fluctuations in biological uptake and release of nutrients, nutrients could be increasing in Snap

Lake without showing a strong correlation with conductivity. These nutrients were selected because results can be compared to a WQG, an EAR prediction, or both.

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. In addition, plots of pH were reviewed for both potential decreasing and increasing trends.

The whole-lake average concentrations of two key parameters, TDS and nitrate, were plotted over the 2004 to 2012 period and compared to TDS and nitrate predictions in the relevant EAR predictions for the equivalent time period (i.e., Years 1 to 7 of the Mine operation). 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.

Water quality data for Northeast Lake and Lake 13, the current and provisional reference lakes, respectively, were visually reviewed for temporal trends and compared to the water quality data from Snap Lake. Notable changes in water quality are not expected in Northeast Lake or Lake 13 and, 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. In addition to determining effects due to the Mine, these plots were used to determine whether Lake 13 would be considered an acceptable reference lake for the water quality component of the AEMP (Section 12.3).

Comparison to Environmental Assessment Report Predictions and 2011 Water Licence Renewal Application Predictions

Maximum observed concentrations of water quality parameters in Snap Lake in 2012 were compared against maximum whole-lake average concentrations predicted in both the EAR (De Beers 2002) and the recent modelling update for the 2011 Water Licence Renewal Application (De Beers 2011b). If concentrations were above the relevant predictions, an attempt was made to determine the source of the increase and the relevance of the elevated concentrations to aquatic biota. Where appropriate, this involved additional comparison of average conditions to relevant EAR predictions as well as comparison of the observed concentrations to relevant AEMP benchmarks.

Seasonal Kendall Test

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 (standard deviation) 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 2012 (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: SNAP13 and SNP 02-20e, SNAP05, SNAP09, SNAP08, SNAP02, and SNAP02A.

Five parameters (i.e., TP, calculated total nitrogen [TN], calculated TDS, total molybdenum, and total strontium) were identified to represent the major parameter groups: major ions, nutrients, and metals. The calculated TN concentrations were determined from the sum of the total Kjeldahl nitrogen (TKN) concentration and the combined nitrate and nitrite concentration. Total manganese, cadmium, fluoride, and antimony were also assessed using the Seasonal Kendall test to support comparisons against EAR benchmarks or WQGs (Section 3.4).

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. As well, a series of figures showing the plume at snapshots through time was 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 an inverse distance weighted method of interpolation, which estimates conductivity values between sampling stations by averaging conductivity in the neighbourhood of each cell, which was set to the nearest 12 sampling stations in Snap Lake. The closer a sampling station is to the centre of the cell being estimated, the more influence, or weight it has on the averaging process. The maps presenting near-surface and near-bottom conductivity values were based on the single field conductivity measured nearest to the surface and bottom, respectively, at each station.

Field conductivity profiles were not available for all AEMP stations in September because the field conductivity data collected at the diffuser stations (SNAP03, SNAP12, SNAP26, and SNAP28) during two days of the September program were found to be unreliable (Appendix 3A). In the

September vertical plot, in addition to the available valid field conductivity profiles, laboratory conductivity values were used to roughly show the vertical profiles at the diffuser stations, where samples were collected at the bottom, mid-depth, and surface of the lake. The quality issues found in the September 2012 field conductivity data are discussed in more 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, near-field, mid-field, far-field, and northwest arm) and from Northeast Lake and Lake 13 were separated by season (i.e., open-water and ice-cover). Results from the ice-covered season involved data collected between January and May 2012, and open-water results involved data collected between July and September 2012.

Downstream Extent of Treated Effluent Plume

Water quality data for the AEMP downstream station, KING01, were reviewed to identify potential changes in water quality at a station located 25 km downstream of Snap Lake. Temporal patterns in TDS and conductivity were reviewed at KING01 to identify trends in TDS. If an increasing trend was detected at KING01, an evaluation of the potential for increases in Mine-related parameters to cause changes in water quality at KING01 would be recommended. The annual water quality results at KING01 were compared to AEMP benchmarks, and historical data. A Seasonal Kendall test for temporal trends was completed.

In addition to the KING01 station, a Downstream Lakes Special Study was conducted in three lakes (Lake 1, Lake 2, and Lac Capot Blanc) immediately downstream of Snap Lake to delineate the spatial extent of the treated effluent plume and assess current conditions. Details of the Downstream Lakes Special Study are described in Section 12.4.

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 reviewed to identify any changes in pH and total alkalinity due to potential acid deposition resulting from Mine emissions. 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?

Water quality parameters in Snap Lake were predicted to remain below drinking WQGs. The AEMP Response Framework outlined in De Beers (2012c) provides a systematic approach for responding to the findings of the AEMP. The level of change in Snap Lake that is not acceptable, based on the EAR, would occur when the water might not be safe to drink. Therefore, water quality data collected from various locations in Snap Lake as part of the AEMP, as well as information collected from the water intake station (i.e., SNP 02-15), were compared against Canadian drinking WQGs (Health Canada 2012). Canadian drinking WQGs that are health based are reported as maximum acceptable concentrations (MAC). Those WQGs 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 do influence a user's perception of water drinkability. However, these objectives are not an indication of adverse effects to human health. The AEMP Response Framework indicates that action levels for drinking water exclude considerations of coliforms, which would be removed through disinfection, and aesthetic objectives. Thus, although the 2012 water quality data were compared to both the relevant MAC and aesthetic objectives, only the results from the comparison to MACs were discussed in detail.

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 Quality Assurance and Quality Control Procedures and Results for the Water Quality Program (Appendix 3A). The results of the 2012 QA/QC program are summarized below.

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 2012, less than 1% of the laboratory data were invalidated and 26% were qualified. Data that were invalidated were flagged with an 'X' in the De Beers Environmental Database and were not used in the analyses. 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 10%, 11%, and 5% 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 2% for oil and grease to 96% for laboratory pH. Nitrite and nitrate often exceeded warning holding times (92% and 96%, respectively).

Nineteen parameters were qualified due to detectable concentrations in the QC blanks that were near concentrations measured in the lake; results of thirteen of these parameters were classified as having a high frequency of detectable concentrations of these parameters in the blanks. These parameters were reactive silica, total organic carbon and six metals (total aluminum, antimony, boron, cadmium (Alberta Innovates), copper, and zinc; and, dissolved aluminum, antimony, boron, copper and zinc).

3.3.2.2 Invalidated Data

Field Data

Approximately 2% of the field data were invalidated because:

- a field probe used to profile the physicochemical characteristics of the water column was assumed, based on the anomalous results it provided, to be near the sediment boundary or submerged in lake bed sediment;
- a dissolved oxygen saturation value was inconsistent with the measured dissolved oxygen value;
- a bottom pH value was inconsistent with the other pH values measured through the water column; and,
- field conductivity measurements were inconsistent with spatial patterns typically observed in Snap Lake and were different from laboratory conductivity results from the same stations.

Laboratory Data

Specific anomalous data points were also removed, based on the criteria outlined in Appendix 3A, Section 3A.1.1.2.4. In 2012, the case-by-case values that were invalidated were:

- conventional parameters: TSS and turbidity collected at IL4 on August 28, 2012;
- ions: reactive silica collected at the diffuser stations on May 13, 2012, and;
- total metals: titanium from SNAP07 collected on February 17, 2012.

These invalidated data were removed because they were considered unusually high results. In each case, the concentrations were more than ten times higher than the average concentrations measured in Snap Lake during the 2012 sampling period. Additional detail is provided in Appendix 3A.

Less than 1% of the ALS results for turbidity and pH were invalidated because the sample holding times had expired, although most of the results that exceeded hold 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 being 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 quality control criteria was low compared to the total number of parameters analyzed. Therefore, the quality of water quality data collected during the 2012 AEMP was considered acceptable and adequate to address the objectives of the monitoring.

3.3.2.3 Calculated Total Dissolved Solids

Calculated TDS is based on the major inorganic ions, measured in mg/L, which could measurably contribute to TDS values in Snap Lake. In 2012, the calculated TDS equation used by ALS was inconsistent with Standard Methods (APHA 2005), past practice (De Beers 2011c), and specific project documentation and approval (De Beers 2005b,c; DFO 2006). The equation included extra parameters, specifically ammonium, iron, manganese, aluminium, copper, zinc, and total organic carbon. The inclusion of the additional parameters, particularly TOC and ammonium, introduced potential positive bias in the lake data. While the bias had minimal effects on conclusions regarding lake dynamics or on temporal trends and spatial patterns, the discrepancy was addressed to provide for consistent and technically defensible data between years. Total dissolved solids values were calculated on an interim basis, using the De Beers Environmental

Database; the interim TDS values are used in this report until ALS re-issues their final revised results. As such, three versions of TDS are presented:

- TDS measured gravimetrically;
- TDS calculated (Lab) [TDS provided by the laboratory, but calculated using an equation not consistent with Standard Methods (APHA 2005)]; and,
- TDS calculated (Standard Methods) [TDS calculated using the equation consistent with Standard Methods].

For the 2012 AEMP, calculated TDS (Standard Methods) was used in all assessments, including plots, calculations, and trend analyses. Further information on calculated TDS is provided in Appendix 3A

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 2012. 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 shallow, with a mean depth of approximately 5 m, and is well-mixed during open-water conditions, with the exception of one deeper area in the northwest arm (greater than 20 m deep), which thermally stratifies in the summer. During ice-covered conditions, limited mixing occurs in Snap Lake. Snap Lake is clear, as indicated by a Secchi depth equal to 6 to 7 m, and has neutral to slightly acidic pH.

Concentrations of DO during ice-covered conditions in Snap Lake tend to be near saturation at the surface, immediately under the ice, and decrease 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 2012, alkalinity in Snap Lake ranged from 7 to 32 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.

Total hardness has increased in Snap Lake since discharge of treated effluent began, thereby lowering the potential for metals toxicity (Chapman 2008). Concentrations increased from 10 mg/L in 2004 to 155 mg/L in 2012 at the outlet of Snap Lake (SNAP08). Major ions and TDS concentrations in Snap Lake were low during baseline conditions, but are increasing as a result of the discharge of treated effluent. Nitrogen and several metals, which are present in elevated concentrations in the discharge, are also 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.

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 the 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 been increasing 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 body 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 body of Snap Lake relative to phosphorus, phosphorus continues to be the limiting nutrient.

Metals concentrations in Snap Lake remained below AEMP benchmarks in 2012. Concentrations of eight metals have increased in Snap Lake: strontium, boron, lithium, barium, 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 because the northwest arm is isolated from the discharge 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 2012, inputs to Snap Lake from Mine-related activities included treated effluent discharge through the diffuser and uncontrolled runoff. The term “treated effluent” refers to combined treated water from the WTP and TWTP, discharged through the diffuser. The diffuser does not influence total loadings to Snap Lake or lake-wide changes in water quality. The 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 itself. 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. Because 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 2012 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 2013a).

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, all 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 minewater effluent. All domestic waste water was treated and routed through the WTP, with the exception of a one-day discharge from the domestic wastewater treatment plant to the wetlands near the northwest arm in 2009. No discharges from the domestic wastewater treatment plant to the wetlands occurred in 2010 or 2011.

- 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.
- In 2012, most of the treated effluent was routed through the WTP, with smaller volumes routed through the temporary WTP until March.
- In March 2012, the treated effluent from the TWTP was redirected through the WTP to the permanent diffuser, after the junction of the water management line from the TWTP was relocated upstream of the WTP.
- During the spring freshet of 2012, a temporary floating diffuser was installed on the ice near the permanent diffuser in accordance with the approved Freshet Water Management Plan (De Beers 2012a). Treated effluent was discharged through both the temporary floating diffuser and the permanent diffuser between May 20, 2012 and June 5, 2012. After June 5, 2012, all treated effluent flows in 2012 were discharged to Snap Lake through the permanent diffuser.

Quantity

Approximately 10 million cubic metres (Mm^3) of treated effluent was discharged from the WTP and TWTP into Snap Lake during the 2012 AEMP reporting year (Table 3-5), which is approximately 12% of the volume of Snap Lake. The discharge volume in 2012 was approximately 27% higher than in 2011. The maximum monthly discharge (i.e., 1.05 Mm^3) was reported in May 2012 (Table 3-5 and Figure 3-4), when the temporary floating diffuser was constructed and placed on the ice in accordance with the approved Freshet Water Management Plan (Section 2; De Beers 2012a).

The WTP operated continuously from November 2011 to October 2012. In March 2012, the pipe from the temporary WTP, which previously joined the pipe from the WTP downstream of the permanent WTP, was relocated to upstream of the permanent WTP. The treated effluent from the TWTP was redirected through the WTP and discharged through the permanent diffuser. Therefore, the volume of treated effluent discharge from the TWTP was included in the discharge volume from the WTP, and reported together in the monthly SNP report from March 2012 onwards.

Table 3-5 Water Treatment Plant Discharge Summary, November 2011 to October 2012

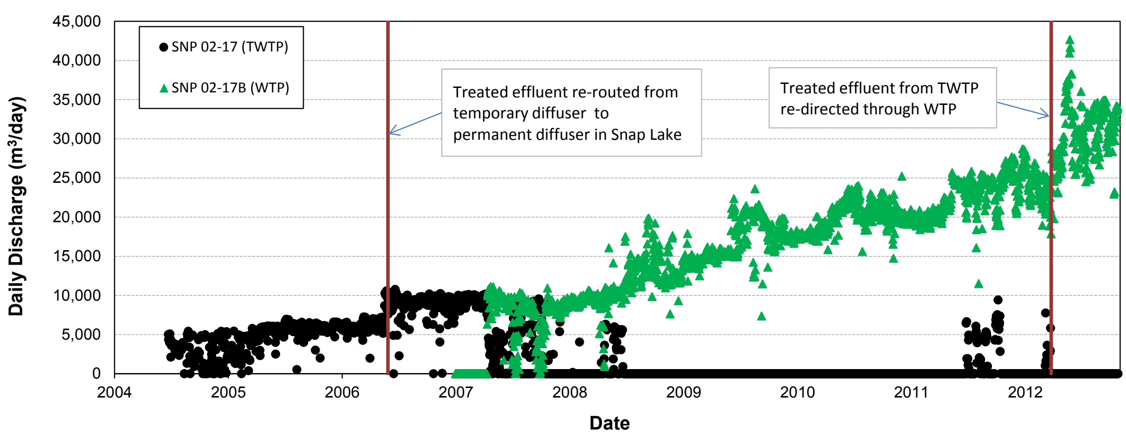
Month and Year	Daily Minimum Flow (m^3/d)	Daily Average Flow (m^3/d)	Daily Maximum Flow (m^3/d)	Total Discharge[(a)] (m^3)
November 2011	24,564	25,976	27,690	779,288
December 2011	20,451	24,544	28,709	760,873
January 2012	21,478	24,927	28,613	772,751
February 2012	21,054	24,259	26,895	703,521
March 2012	17,845	23,510	28,133	728,824
April 2012	19,805	26,252	28,770	787,545
May 2012	25,530	33,780	42,657	1,047,178
June 2012	25,434	30,796	38,264	923,873
July 2012	24,528	30,230	36,032	937,132
August 2012	24,807	31,075	34,626	963,320
September 2012	26,660	32,369	34,956	971,076
October 2012	22,941	31,110	34,206	964,395
Total				**10,339,776[(b)]**

(a) Total discharge represents the total monthly discharge from both the temporary (SNP 02-17) and permanent (SNP 02-17B) water treatment plants.

(b) Total represents the total annual discharge from November 1, 2011 to October 31, 2012 for the temporary (SNP 02-17) and permanent (SNP 02-17B) water treatment plants.

m^3/d = cubic metres per day; m^3 = cubic metres.

Figure 3-4 Treated Effluent Discharge Rate to Snap Lake, 2004 to October 2012



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

Quality

Comparisons to Water Licence Limits

Parameters with Water Licence limits in effect in 2011/2012 were TSS, nitrogen compounds (ammonia, nitrate and nitrite), chloride, sulphate, metals (aluminum, cadmium, chromium, copper, lead, nickel and zinc), and a metalloid (arsenic) (Table 3-6). The Water Licence specifies both a “maximum concentration in any grab sample” and an “average monthly limit” (AML). An AML is the concentration that cannot be exceeded, determined by averaging the analytical results of six consecutive samples collected at 6-day intervals over a 30-day period. For parameters measured every six days (i.e., physical parameters, major ions, nutrients) a 30-day moving average was calculated for comparison. For metals, which are analyzed approximately once per month, a monthly average value was used.

In addition, the pH level is to be maintained within the range of 6 to 9 pH units and TP has an annual load limit of 256 kg/y. All of the aforementioned limits apply at end-of-pipe. The Water Licence also specifies an in-lake limit for whole-lake average TDS concentration of 350 mg/L.

Table 3-6 Water Licence Limits for Treated Effluent

Parameter	Maximum Concentration of Any Grab Sample (mg/L)		Average Monthly Limit (mg/L)		Average Annual Loading (kg/year)	
	2011[(a)]	2012[(b)]	2011[(a)]	2012[(b)]	2011[(a)]	2012[(b)]
Total Suspended Solids	14	14	7	7	-	-
Ammonia, as N	20	20	-	10	187,000	187,000
Total Phosphorus, as P	-	-	-	-	-	256
Nitrite, as N	2	1	1	0.5	-	-
Nitrate, as N	56	44	28	22	219,000	219,000
Chloride	-	620	-	310	-	-
Sulphate	-	150	-	-	-	-
Aluminum	2	0.2	1	0.1	-	-
Arsenic	0.04	0.014	0.02	0.007	-	-
Cadmium	0.002	-	0.001	-	-	-
Chromium	0.04	0.02	0.02	0.01	-	-
Copper	0.02	0.006	0.01	0.003	-	-
Lead	0.009	0.01	0.005	0.005	-	-
Nickel	0.1	0.1	0.05	0.05	-	-
Zinc	0.02	0.02	0.01	0.01	-	-

(a) Water Licence MVLWB (2004).
(b) 2012 Water Licence MVLWB (2012), effective on June 14, 2012.
- = limit not specified; mg/L = milligrams per litre; kg/year = kilograms per year.

Concentrations in treated effluent remained below the maximum allowable concentration in grab samples of treated effluent for most parameters between November 2011 and October 2012.

Exceptions were TSS on June 1 and August 1, 2012, and total aluminum measured on August 1, 2012 (Table 3-7).

The elevated TSS and total aluminum concentrations on August 1, 2012 were a result of an unexpected release of approximately 300 cubic metres (m^3) of turbid water from the WTP. The incident was reported and filed as Spill 12-314. Root cause and follow-up actions for the unexpected release and the Water Licence limit exceedances were provided in De Beers (2012d).

Table 3-7 Results Above Water Licence Criteria

Station	Sample Control Number	Sample Date	Parameter	Unit	Result	Limit – Maximum Concentration of Any Grab Sample[(a)]
SNP 02-17B	2012-1103	June 1, 2012	Total Suspended Solids	mg/L	18	14
SNP 02-17B	2012-1413	August 1, 2012[(b)]	Total Suspended Solids	mg/L	124	14
SNP 02-17B	2012-1413	August 1, 2012[(b)]	Total Aluminum	mg/L	0.234	0.2

(a) Maximum concentration of any grab sample = the concentration of any parameter listed in the Water Licence that cannot be exceeded in any grab sample collected at the final point of discharge (MVLWB 2004, 2012).

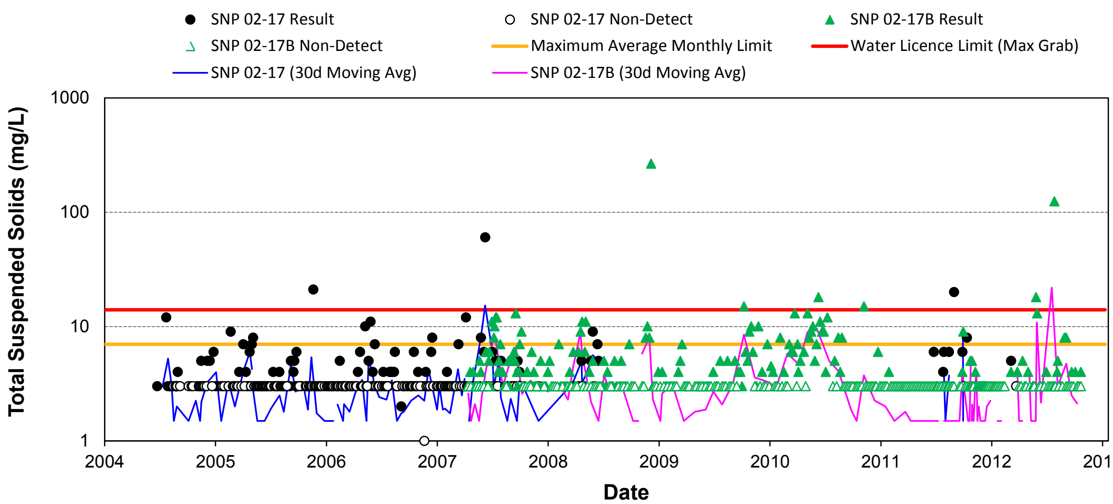
(b) Elevated concentrations related to an unexpected release of turbid water (i.e., Spill 12-314).

mg/L = milligrams per litre; SNP = Surveillance Network Program.

Temporal plots of concentrations and loadings for parameters above Water Licence limits are illustrated in Figures 3-5 to 3-6; plots for remaining parameters are provided in Appendix A4. Values measured below detection limits (DL) were plotted at the DL; values set at half the DL were used to calculate 30-day moving and monthly averages.

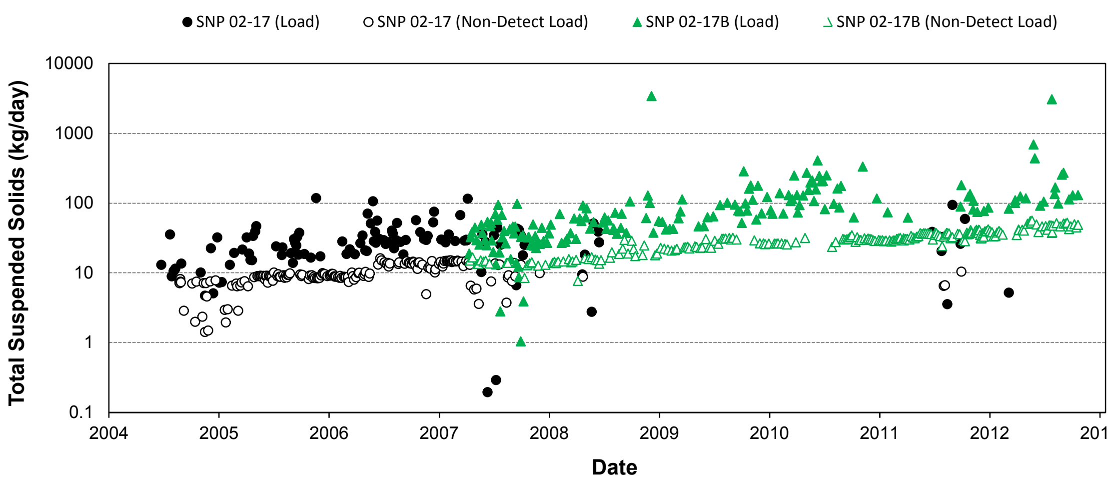
Figure 3-5 Total Suspended Solids Concentrations in Treated Effluent, 2004 to October 2012

a. Concentration and Moving Average



Non-Detect = values reported as less than the detection limit; 30d Moving Avg = 30-day moving average; Max Grab = maximum allowable concentration in any grab sample; 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.

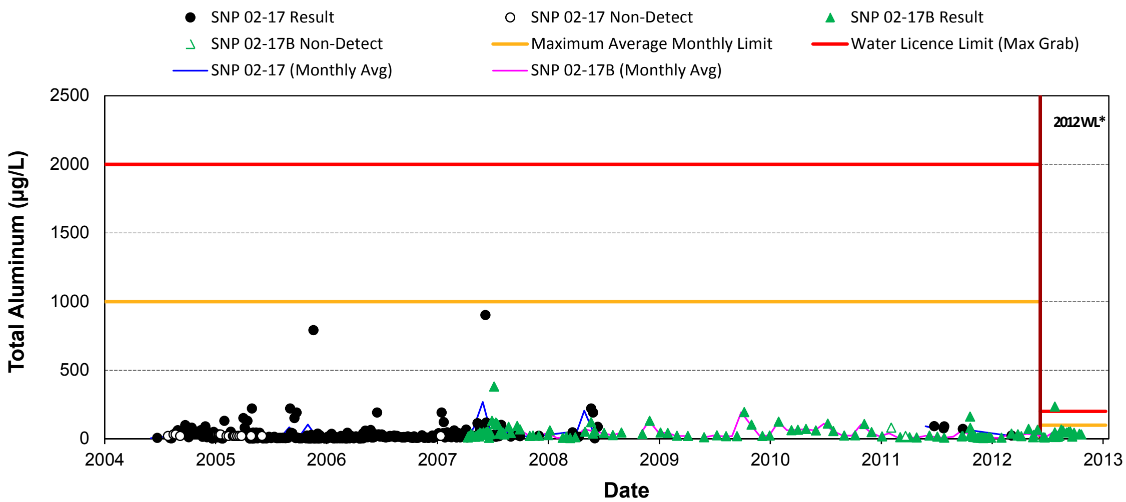
b. Loading



Non-Detect = values reported as less than the detection limit; 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 Aluminum Concentrations in Treated Effluent, 2004 to October 2012

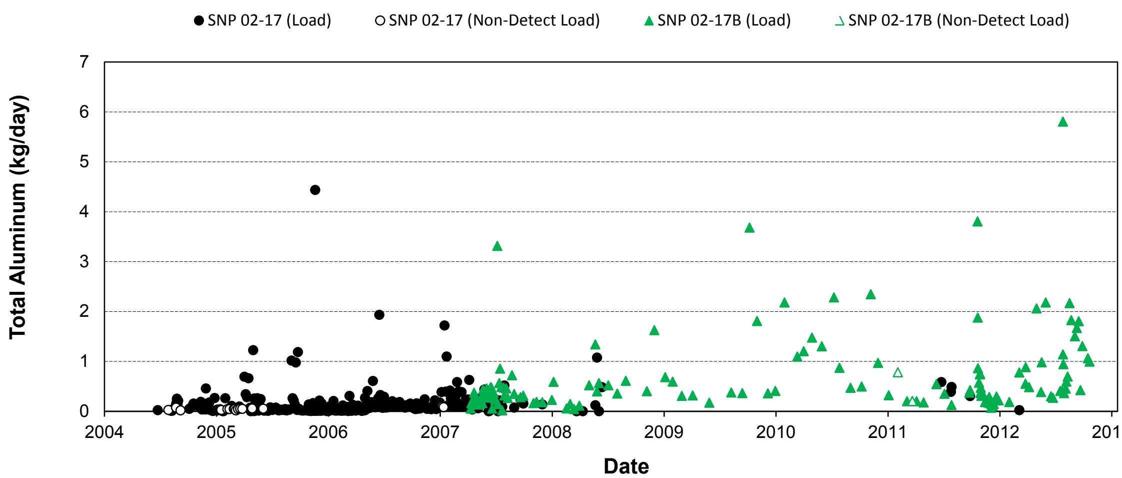
a. Concentration and Moving Average



* The Water Licence limits (maximum concentration of any grab sample and maximum average monthly limit) for total aluminum were lowered from 2,000 µg/L and 1,000 µg/L to 200 µg/L and 100 µg/L, respectively, when the new Water Licence came into effect on June 14, 2012: MV2011L2-0004 (MVLWB 2012).

Non-Detect = values reported as less than the detection limit; Monthly Avg = monthly average; SNP 02-17 = treated effluent from the temporary water treatment plant; SNP 02-17B = treated effluent from the permanent water treatment plant; Max Grab = maximum allowable concentration in any grab sample; µg/L = micrograms per litre.

b. Loading



Non-Detect = values reported as less than the detection limit; 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.

Signature Parameters in Treated Effluent from the Mine

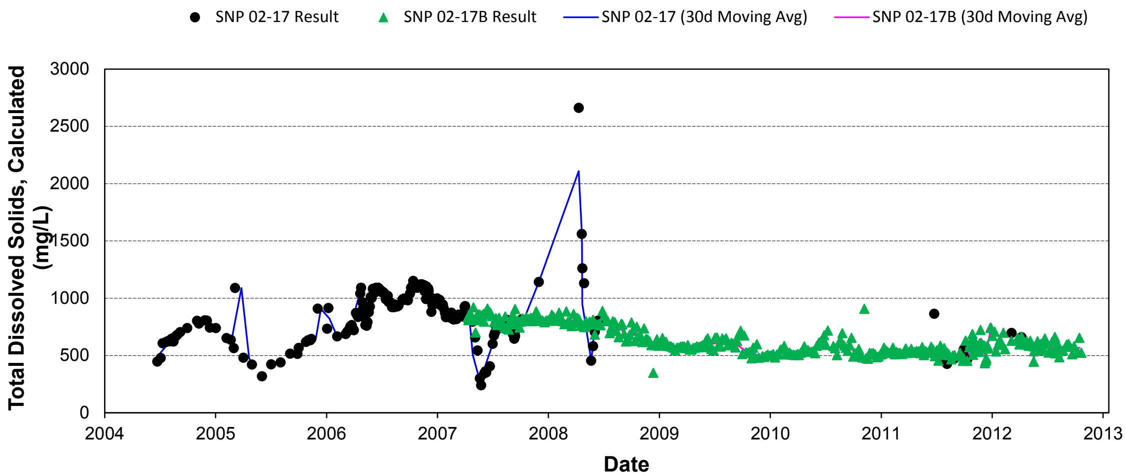
Chemical signatures in treated effluent from the Mine are:

- TDS and its component ions (i.e., calcium, chloride, fluoride, magnesium, nitrate and nitrite, potassium, sodium, sulphate, and total alkalinity);
- nitrogen nutrients (e.g., ammonia, nitrate, and nitrite); and,
- some metals (e.g., barium, boron, lithium, molybdenum, nickel, rubidium, strontium and uranium).

Metals and TDS originate from extraction of deep groundwater during the mining process, whereas the nitrogen nutrients are present in treated domestic wastewater and are a by-product from ammonium nitrate fuel oil (ANFO) use during blasting. Concentrations for nine signature parameters (TDS, calcium, chloride, magnesium, sulphate, sodium, barium, lithium, and strontium) in the treated effluent from Mine remained relatively consistent in 2012 (Figure 3-7 and Appendix 3D). Fluoride concentrations in the treated effluent have consistently decreased since 2004 (Figure 3-8). Seasonal patterns were evident for nitrogen parameters (nitrate, ammonia, and nitrite), with higher concentrations occurring during the spring/summer (Figure 3-9). No clear increasing or decreasing trend in concentrations could be identified for some parameters (pH, turbidity, bicarbonate, sulphate, TKN, and TP). However, loadings to Snap Lake from the treated effluent have increased over time due to increased daily discharge volumes since 2004 (Figure 3-4).

Figure 3-7 Total Dissolved Solids Concentrations in Treated Effluent, 2004 to October 2012

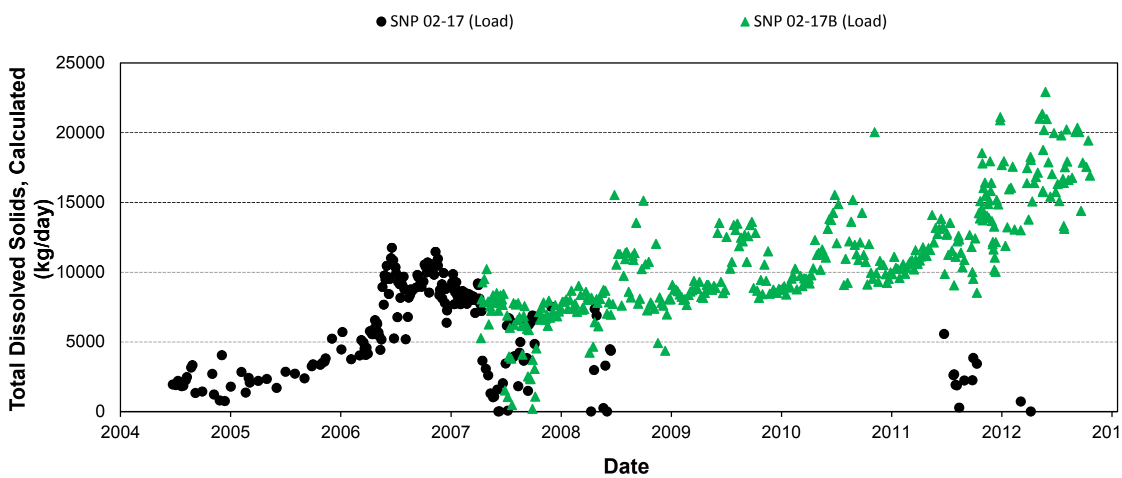
a. Concentration and Moving Average



Note: Total dissolved solids (TDS) calculation formula was updated in January 2012. The 2012 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).

30d Moving Avg = 30-day moving average; 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.

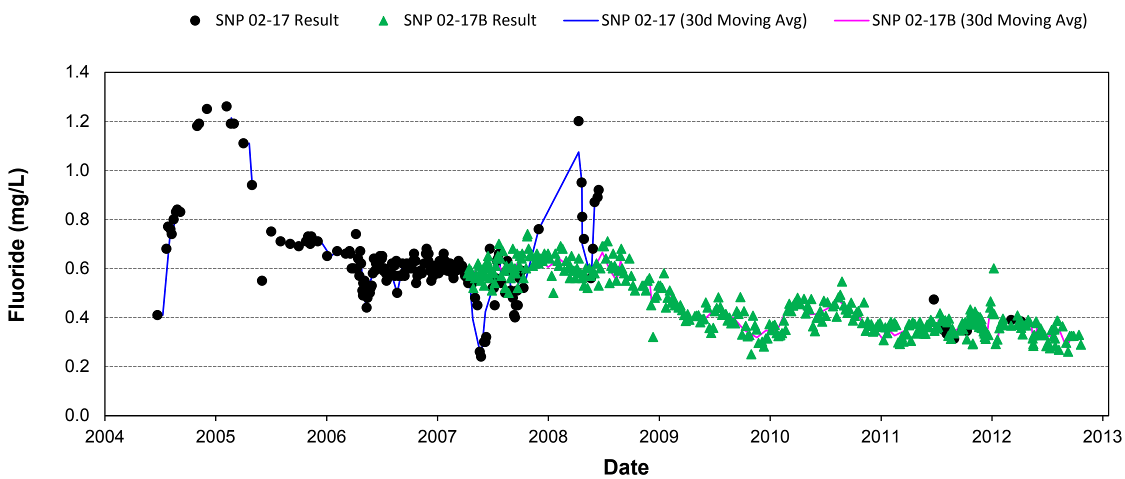
b. Loading



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.

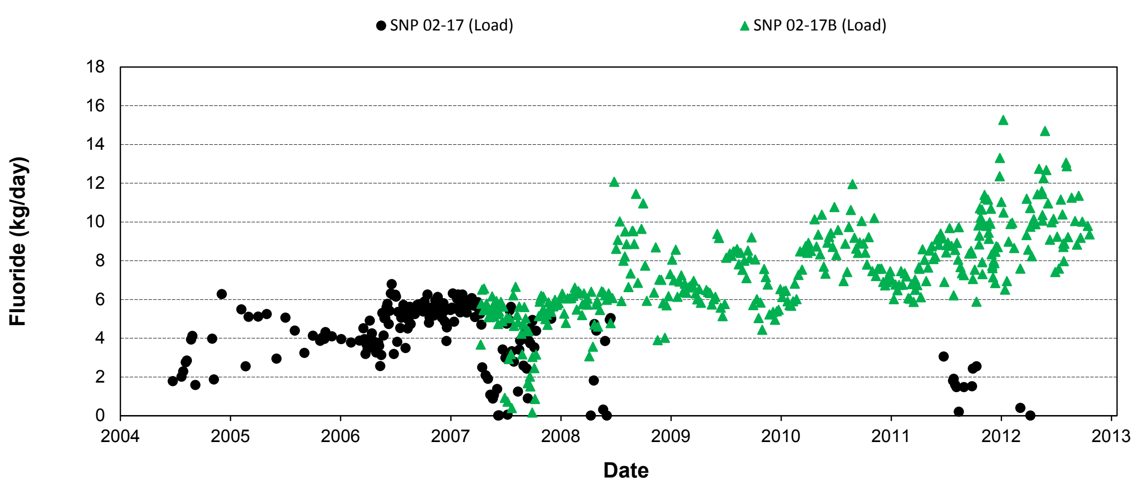
Figure 3-8 Fluoride Concentrations in Treated Effluent, 2004 to October 2012

a. Concentration and Moving Average



30d Moving Avg = 30-day moving average; 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.

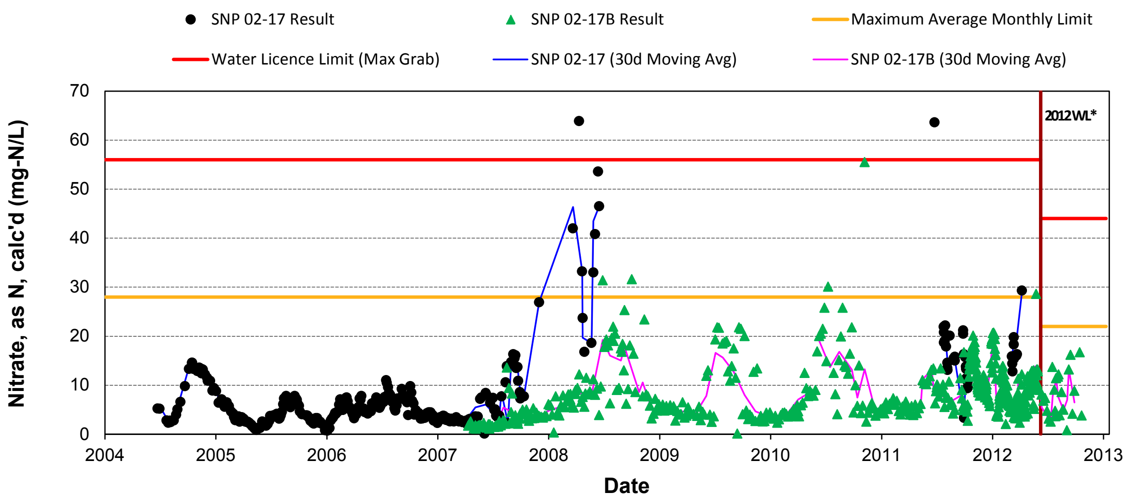
b. Loading



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.

Figure 3-9 Nitrate Concentrations in Treated Effluent, 2004 to October 2012

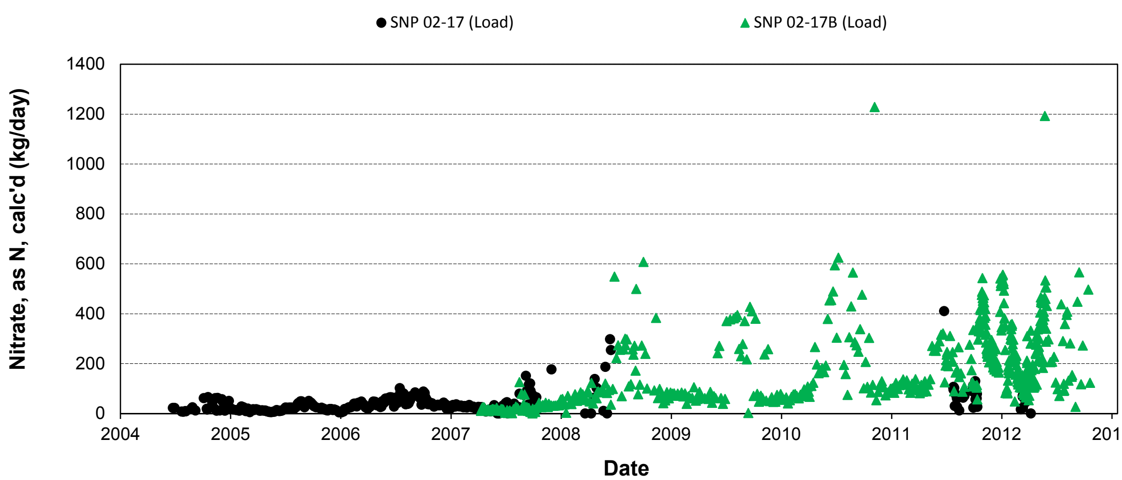
a. Concentration and Moving Average



*The Water Licence limit of maximum concentration of any grab sample and maximum average monthly limit for total aluminum were lowered from 56 mg-N/L and 28 mg-N/L to 44 mg-N/L and 22 mg-N/L, respectively, when the new Water Licence came into effect on June 14, 2012: MV2011L2-0004 (MVLWB 2012).

Non-Detect = values reported as less than the detection limit; Monthly Avg = monthly average; SNP 02-17 = treated effluent from the temporary water treatment plant; SNP 02-17B = treated effluent from the permanent water treatment plant; Max Grab = maximum allowable concentration in any grab sample; mg-N/L = milligrams as nitrogen per litre.

b. Loading



Non-Detect = values reported as less than the detection limit; 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 Load to Snap Lake

The average flow-weighted concentrations of TP in the WTP and TWTP discharges were 0.0033 mg/L and 0.0065 mg/L, respectively (Tables 3-8 and 3-9). These values resulted in an estimated total load of TP to Snap Lake of 67 kilograms (kg) between November 1, 2011 and October 31, 2012 (Table 3-10). The Water Licence specifies an annual TP load of less than 256 kg/year to Snap Lake (MVLWB 2012); thus, loadings were well below the Water Licence limits.

Table 3-8 Daily Phosphorus Loadings and Flow-Weighted Average Concentrations from the Temporary Water Treatment Plant, SNP 02-17, November 1, 2011 to October 31, 2012

Date	Total Phosphorus (mg/L)	Average Daily Discharge (m^3)	Average Daily Phosphorus Load (g)
11-Mar-12	0.0023	1,038	2
18-Mar-12	0.0029	1,472	4
26-Mar-12	0.0038	2,772	11
Total		**5,282[(a)]**	**17[(b)]**
Weighted average total phosphorus concentration:			= 17 g / 5,282 m^3 = 0.0033 mg/L

Note: Date defined as dd-mmm-yy where d is day, m is month, and y is year; in March 2012 the treated effluent from the temporary water treatment plant was redirected through the permanent water treatment plant.

(a) Total volume during sampling period.

(b) Total loading during sampling period.

mg/L = milligrams per litre; m^3 = cubic metres; g = gram.

Table 3-9 Daily Phosphorus Loadings and Flow-Weighted Average Concentrations from the Permanent Water Treatment Plant, SNP 02-17B, November 1, 2011 to October 31, 2012

Dates	Total Phosphorus (mg/L)	Average Daily Discharge (m^3)	Average Daily Phosphorus Load (g)
1-Nov-11	0.0029	25,560	74
2-Nov-11	0.0024	26,870	64
3-Nov-11	0.0020	26,990	54
4-Nov-11	0.0030	27,000	81
5-Nov-11	0.0036	27,200	98
6-Nov-11	0.0034	24,920	85
7-Nov-11	0.0018	25,740	46
8-Nov-11	0.0019	25,270	48
13-Nov-11	0.0015	26,870	40
14-Nov-11	0.0017	26,780	46
20-Nov-11	0.0019	26,270	50
22-Nov-11	<0.001	24,980	12
24-Nov-11	<0.001	27,690	14
26-Nov-11	<0.001	26,151	13
27-Nov-11	0.0025	24,940	62
30-Nov-11	0.0033	24,880	82
2-Dec-11	0.0028	27,585	77
4-Dec-11	0.0049	27,246	134
6-Dec-11	0.0038	23,321	89
8-Dec-11	0.0033	21,002	69
9-Dec-11	0.0023	20,753	48
11-Dec-11	0.0017	20,836	35
13-Dec-11	0.0017	22,683	39
14-Dec-11	0.0033	23,534	78
15-Dec-11	0.0033	24,613	81
18-Dec-11	0.0037	21,953	81
19-Dec-11	<0.001	25,626	13
22-Dec-11	0.0035	26,023	91
27-Dec-11	0.0042	26,529	111
2-Jan-12	0.0063	28,080	177
3-Jan-12	0.0045	28,613	129
3-Jan-12	0.0040	28,613	114
8-Jan-12	0.0037	26,407	98
14-Jan-12	0.0100	25,441	254
15-Jan-12	0.0028	25,401	71
20-Jan-12	<0.001	22,276	11
26-Jan-12	0.0047	22,418	105
1-Feb-12	0.0054	25,232	136
7-Feb-12	0.0063	26,388	166
13-Feb-12	0.0052	25,220	131
19-Feb-12	0.0044	23,515	103

Table 3-9 Daily Phosphorus Loadings and Flow-Weighted Average Concentrations from the Permanent Water Treatment Plant, SNP 02-17B, November 1, 2011 to October 31, 2012

Dates	Total Phosphorus (mg/L)	Average Daily Discharge (m^3)	Average Daily Phosphorus Load (g)
25-Feb-12	0.0032	23,376	75
2-Mar-12	0.0119	18,957	226
11-Mar-12	0.0066	20,745	137
14-Mar-12	0.0046	24,147	111
20-Mar-12	0.0034	22,539	77
26-Mar-12	0.0034	22,529	77
31-Mar-12	0.0069	25,140	173
1-Apr-12	0.0076	27,438	209
6-Apr-12	0.0052	23,145	120
12-Apr-12	0.0032	27,754	89
13-Apr-12	0.0014	27,232	38
18-Apr-12	0.0056	24,856	139
24-Apr-12	0.0047	27,646	130
30-Apr-12	0.0056	26,914	151
6-May-12	0.0099	29,308	290
12-May-12	0.0095	33,538	319
18-May-12	0.0113	36,908	417
21-May-12	0.0018	36,932	66
23-May-12	0.0030	35,441	106
23-May-12	0.0035	35,441	124
24-May-12	0.0082	35,092	288
27-May-12	0.0094	37,367	351
1-Jun-12	0.0240	38,257	918
5-Jun-12	0.0090	33,641	303
11-Jun-12	0.0050	32,910	165
17-Jun-12	0.0067	25,598	172
23-Jun-12	0.0080	30,524	244
29-Jun-12	0.0062	30,884	191
5-Jul-12	0.0046	26,202	121
11-Jul-12	0.0094	31,079	292
17-Jul-12	0.0062	27,704	172
23-Jul-12	0.0056	31,968	179
29-Jul-12	0.0058	31,972	185
1-Aug-12	0.1240	24,807	3,076
1-Aug-12	0.0062	24,807	154
2-Aug-12	0.0058	25,407	147
4-Aug-12	0.0066	31,482	208
10-Aug-12	0.0123	33,742	415
12-Aug-12	0.0095	33,436	318
16-Aug-12	0.0090	34,394	310
22-Aug-12	0.0104	32,722	340

Table 3-9 Daily Phosphorus Loadings and Flow-Weighted Average Concentrations from the Permanent Water Treatment Plant, SNP 02-17B, November 1, 2011 to October 31, 2012

Dates	Total Phosphorus (mg/L)	Average Daily Discharge (m^3)	Average Daily Phosphorus Load (g)
28-Aug-12	0.0074	30,914	229
3-Sep-12	0.0043	31,728	136
9-Sep-12	0.0055	33,814	186
15-Sep-12	0.0062	33,957	211
21-Sep-12	0.0072	34,920	251
27-Sep-12	0.0060	28,232	169
3-Oct-12	0.0045	30,798	139
9-Oct-12	0.0033	32,280	107
15-Oct-12	0.0049	33,095	162
21-Oct-12	0.0049	29,762	146
27-Oct-12	0.0042	32,380	136
Total		**2,593,280[(a)]**	**16,806[(b)]**
Weighted average total phosphorus concentration		= 16,806 g / 2,593,280 m^3	= 0.0065 mg/L

Note: Date defined as dd-mmm-yy, where d = day, m = month, and y = year.

Average daily phosphorus load was calculated using data at the same precision level provided by the laboratory; tabulated data were rounded.

(a) Total volume during sampling period.

(b) Total loading during sampling period.

mg/L = milligrams per litre; m^3 = cubic metres; g = gram; < = less than.

Table 3-10 Total Phosphorus Load Discharged From the Water Treatment Plants, November 1, 2011 to October 31, 2012

Station	Count	Min	Flow-Weighted Average [(a)] (mg/L)	Max	Total Volume (m^3)	Total Volume (L)	Total Phosphorus (kg)
Treated Effluent Discharged through the Diffuser							
SNP 02-17	3	0.002	0.003	0.004	29,563	29,563,000	0.1
SNP 02-17B	93	<0.001	0.006	0.124	10,310,206	10,310,206,181	66.8
Total Phosphorus Load Discharged to Snap Lake from November 2011 to October 2012 (kg)							66.9

(a) Flow-weighted average total phosphorus concentration from the temporary (SNP 02-17) and permanent (SNP 02-17B) WTPs as presented in Table 3-8 and Table 3-9, respectively.

SNP = Surveillance Network Program; WTP = water treatment plant; min = minimum; max = maximum; kg = kilogram; L = litre; m^3 = cubic metres; mg/L = milligrams per litre; <= less than.

Toxicity of Discharge

Acute and chronic toxicity tests were conducted on treated effluent samples on a quarterly basis. Details of the toxicity test methods and of the results for the treated effluent samples are provided in Appendix 3E, including graphical summaries of the chronic toxicity data. A summary of the results is provided below.

The 2012 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 2012.

Chronic toxicity was predicted to occur in treated effluent in the EAR (De Beers 2002). In 2012, one treated effluent sample from the permanent WTP showed evidence of chronic toxicity in terms of *Ceriodaphnia dubia* survival but not 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.

The temporal evaluation of chronic toxicity from 2005 to 2012 demonstrated that, although toxicity did occur in chronic tests performed on the treated effluent, it did not show a temporal trend of increasing frequency or severity over time.

Comparisons to EAR Predictions

In general, treated effluent had higher concentrations of major ions, nutrients, and metals compared to Snap Lake. Flow-weighted average concentrations of all water quality parameters measured in the water treatment plant discharges in 2012 were compared to EAR predictions (Table 3-11). The combined flow-weighted average concentrations were below maximum annual average concentrations for treated effluent predicted in the EAR, with the exception of sulphate (Table 3-11). Sulphate is a component of TDS (i.e., approximately 9%), so it will be implicitly considered as part of the ongoing aquatic toxicity testing being conducted to develop an appropriate site-specific, effects-based TDS water quality benchmark. Sulphate was not identified as a key parameter during the most recent lake model update (De Beers 2011b) because CCME currently does not recommend a WQG for sulphate. Sulphate will be included in future lake model updates following benchmark development.

Table 3-11 Summary of Treated Effluent Discharge Above EAR Predictions

Parameters	Units	SNP 02-17				SNP 02-17B				Combined				EAR Predictions[a]		
		Min	Flow-Weighted Avg[b]	Max	Count	Min	Flow-Weighted Avg[b]	Max	Count	Min	Flow-Weighted Avg[b]	Max	Count	Max Weekly	Max Average Annual	Avg
Conventional Parameters																
Total Dissolved Solids, calculated[c]	mg/L	681	*685*	701	3	432	5,693	7,443	91	432	570	744	94	1,332	929	592
Total Suspended Solids	mg/L	4	***5***	7	3	<3	3	124	91	<3	3	124	94	5	5	5
Major Ions																
Chloride	mg/L	279	*280*	282	3	166	*246*	335	93	166	*246*	335	96	425	374	237
Magnesium	mg/L	18	*20*	20	3	11	15	23	92	11	15	23	95	25	21	16
Sodium	mg/L	66	*68*	71	3	43	*59*	84	92	43	*59*	84	95	78	69	38
Sulphate	mg/L	64	***70***	74	3	37	***54***	68	92	37	***54***	74	95	46	40	17
Nutrients																
Nitrate, as N, calculated	mg-N/L	16	***17.3***	18.2	3	0.8	*10.2*	21.7	93	0.8	*10.2*	21.7	96	15.8	13.3	5.8
Total Metals																
Cobalt	µg/L	0.7	*0.9*	1.0	2	<0.1	0.4	1.6	59	<0.1	0.4	1.6	61	3.4	3.2	0.6
Manganese	µg/L	95	*102*	104	2	<2	*66*	150	59	<2	*66*	150	61	156	146	30
Nickel	µg/L	16	**17**	17	2	2	11	22	59	2	11	22	61	61	15	14
Strontium	µg/L	1,720	*1,761*	1,870	2	165	*1,563*	2,110	59	165	*1,563*	2,110	61	2,616	2,346	1,501
Uranium	µg/L	0.9	*1.0*	1.1	2	0.5	*0.9*	1.5	59	0.5	*0.9*	1.5	61	17.7	1.2	0.7

Note: *Italics* indicates a concentration above the predicted average; **Bold** indicates a concentration above the predicted maximum annual average; **Bold** and *Italics* indicate a concentration above the predicted weekly maximum average.

(a) EAR Predictions from De Beers (2002)

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

(c) The 2012 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).

Min = minimum; Avg = average; Max. = maximum; 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.

3.4.2.3 Dilution Factors Associated With the Permanent Diffuser

The minimum dilution factors at 200 m away from the diffuser (i.e., the edge of the mixing zone) in February, April, July, and September 2012 were 18, 28, 9, and 37, respectively (Table 3-12). These minimum dilution factors are based on the maximum observed TDS concentrations at the diffuser stations and, therefore, represent the least amount of dilution that was provided by the diffuser based on observed results.

The 2012 dilution factors calculated from observed data collected in February and April were lower than the minimum dilution factor of 34 predicted in the EAR for ice-covered conditions (De Beers 2002). The lower dilution factors in 2012 were attributed to air entrainment observed in the discharge, which resulted in only two or three of the five diffuser ports discharging treated effluent while the other ports discharged air (De Beers 2013a). 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.

Modelling results from the Plume Characterization Study indicated that the existing configuration of the diffuser should provide the dilution predicted in the EAR, as long as air is minimized in the discharge (Golder 2013). De Beers took steps to minimize air in the discharge in September 2012 (De Beers 2012e), which is expected to result in improved performance of the diffuser and therefore higher dilution factors in 2013.

The lowest calculated dilution factors in 2012 occurred during the early open-water season in July, which is consistent with 2011 (De Beers 2012b) but not from 2007 to 2010 (De Beers 2008a, 2009, 2010, 2011a). The lower dilution factors observed in July may be influenced by the presence of stratified conditions at one diffuser station (i.e., SNP 02-20e) for a short time during the early open-water season. The results of the modelling completed for the plume characterization study also indicated that open-water stratified conditions may be the most limiting time for mixing in that area (Golder 2013).

Table 3-12 Dilution Factors for the Permanent Diffuser Based on 2012 Total Dissolved Solids Results

Month	Average Discharge Rate (m^3/d)	Discharge Range (m^3/d)	TDS, Calculated (Standard Methods) Concentrations (mg/L)			Minimum Dilution Factor
			Snap Lake Background[(a)]	Maximum at any Depth at Diffuser Stations	Treated Effluent[(b)]	
February	24,259	Min: 21,054 Max: 26,895	223	245	616	18
April	26,252	Min: 19,805 Max: 28,770	243	256	626	28
July	30,230	Min: 24,528 Max: 36,032	185	226	561	9
September	32,369	Min: 26,660 Max: 34,956	195	205	569	37

(a) Average of TDS concentrations that were collected from the near-field sampling stations in Snap Lake (i.e., SNAP03, SNAP05, SNAP06, SNAP12, SNAP26, and SNAP28) within the 2012 reporting period (January 1, 2012 to September 30, 2012).

(b) Combined flow-weighted average TDS concentration from the temporary (SNP 02-17) and permanent (SNP 02-17B) WTP.

Min = minimum; Max = maximum; m^3/d = cubic metres per day; mg/L = milligrams per litre; TDS = total dissolved solids.

3.4.2.4 Summary of Key Question 1

The volume of daily discharge to Snap Lake has increased since 2004. Loadings of signature parameters of treated effluent from the Mine, which included TDS and its component ions (calcium, chloride, fluoride, magnesium, nitrate, potassium, sodium, and sulphate), nitrogen nutrients (ammonia, nitrate and nitrite), and metals (e.g., barium, boron, lithium, molybdenum, nickel, rubidium, strontium and uranium), have increased due to increases in daily discharge rates. Concentrations in the treated effluent remained below the maximum allowable concentration in any grab sample of treated effluent for most parameters in 2012. Exceptions were two TSS results and one total aluminum concentration.

Flow-weighted average concentrations of sulphate have routinely been above the maximum average annual concentration predicted in the EAR. The 2012 TP loading to Snap Lake from the WTP was 67 kg, which was well below the Water License limit of 256 kg. The 2012 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 2012, one treated effluent sample from the permanent WTP showed evidence of chronic toxicity in terms of *Ceriodaphnia dubia* survival but not 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 data collected in Snap Lake during the 2012 AEMP program were compared against AEMP benchmarks (Table 3-13). The AEMP benchmarks were: CCME (1999) WQGs, and site-specific EAR benchmarks developed for copper, cadmium, and hexavalent chromium (De Beers 2002). Generally, water quality parameters in Snap Lake were below AEMP benchmarks with the exception of chloride, fluoride, and nitrate (Table 3-13). Dissolved oxygen concentrations were, on occasion, below the minimum CCME WQG. For these 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 different areas of the main basin of Snap Lake (i.e., diffuser, near-field, mid-field, and far-field). Concentrations in Northeast Lake and Lake 13 are also presented in Table 3-13 for reference. Comparisons to EAR model predictions (De Beers 2002) and 2011 Water Licence Renewal Application model predictions (De Beers 2011b) are provided in Key Question 3 (Section 3.4.4).

Table 3-13 Comparison of 2012 Snap Lake Water Quality to AEMP Benchmarks

Parameter	Units	AEMP Benchmarks (Protection of Aquatic Life)[a]	Observed Concentrations[b]			
			Type	Snap Lake	Northeast Lake[c]	Lake 13[c]
Field Parameters						
Dissolved Oxygen	mg/L	6.5, 9.5[d]	min and range in average[e]	**1.0** and 9.1 to 14.2	**5.1**	6.7
Conventional Parameters						
Laboratory pH	unitless	6.5 to 9.0	range	6.8 to 7.7	6.8 to 7.3	7.1 to 7.2
Major Ions						
Chloride	mg/L	120	max and range in average[e]	**121** and 84 to 112	1	5
Fluoride	mg/L	0.12	max and range in average[e]	**0.18** and 0.05 to **0.13**	0.08	0.05
Nitrate, as N	mg-N/L	2.93	max and range in average[e]	**3.22** and 1.6 to 2.7	0.03	<0.006
Nitrite, as N	mg-N/L	0.06	max	0.029	0.015	<0.002
Ammonia, as N	mg-N/L	1.0 to 21.6[f]	max	0.32	0.01	<0.005
Total Metals						
Aluminum	µg/L	100[g]	max	15	7	8
Arsenic	µg/L	5	max	0.2	0.1	0.1
Boron	µg/L	1,500	max	53	5	3
Cadmium	µg/L	0.36	max	0.07	0.01	<0.002
Chromium	µg/L	8.9	max	0.3	0.27	<0.06
Hexavalent chromium	µg/L	2.1	max	1.2	<1	<1
Copper	µg/L	7.9	max	0.8	0.6	0.39
Iron	µg/L	300	max	19	12	14
Lead	µg/L	1 to 7[h]	max	0.07	0.03	<0.01
Mercury (Flett)	µg/L	0.026	max	0.0012	0.0007	<0.0005
Molybdenum	µg/L	73	max	1.5	0.06	<0.05
Nickel	µg/L	33.9 to 153[h]	max	2.2	0.4	0.2

Table 3-13 Comparison of 2012 Snap Lake Water Quality to AEMP Benchmarks

Parameter	Units	AEMP Benchmarks (Protection of Aquatic Life)[(a)]	Observed Concentrations[(b)]			
			Type	Snap Lake	Northeast Lake[(c)]	Lake 13[(c)]
Selenium	µg/L	1	max	0.044	<0.04	<0.04
Silver	µg/L	0.1	max	<0.005	<0.005	<0.005
Thallium	µg/L	0.8	max	<0.01	<0.01	<0.01
Uranium	µg/L	15	max	0.2	0.02	0.02
Zinc	µg/L	30	max	9	4	2

Note: Only parameters with AEMP benchmarks are presented in Table 3-13.

(a) AEMP benchmarks are: Water Quality Guidelines (WQGs) from the Canadian Council of Ministers of the Environment (CCME) (1999 with updates to 2012) 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).

(b) Observed concentrations within the 2012 reporting period (January 1, 2012 to September 30, 2012). **Bold** values were above the relevant benchmarks.

(c) Maximum observed concentrations in Northeast Lake and Lake 13 within the 2012 reporting period.

(d) 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.

(e) Range in average = minimum and maximum average concentrations in different areas of the main basin of Snap Lake (i.e., diffuser, near-field, mid-field, and far field).

(f) The ammonia WQG is pH and water temperature dependent. The range of the guideline shown is based on a range of laboratory pH from 6.8 to 7.7 and a range of water temperature from 0.9°C to 18.1°C, which were observed in Snap Lake over the 2012 reporting period. The guideline was calculated based on an individual pH and water temperature for each sample with the final value expressed as ammonia nitrogen.

(g) The aluminum WQG is pH dependent. The guideline shown here is based on a range of pH from 6.8 to 7.7, which was observed in Snap Lake during the 2012 reporting period. The WQG was calculated based on the individual pH for each sample.

(h) The lead and nickel WQGs are hardness dependent. The range of the WQGs shown was based on a range of hardness from 26 to 186 mg/L, which was observed in Snap Lake during the 2012 reporting period. The WQG was calculated based on the individual hardness for each sample.

N = nitrogen; - = not applicable; <= less than; ≤ = less than or equal to; min = minimum; 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.

Chloride

In 2012, concentrations of chloride in Snap Lake were typically below the CCME WQG of 120 mg/L, with the exception of one chloride result of 121 mg/L collected from the diffuser area at station SNP 02-20e. Average chloride concentrations in the different lake areas ranged from 84 mg/L to 112 mg/L and were below the CCME WQG (Table 3-13).

The observed 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). Using the hardness-based formula provided in Elphick et al. (2011), and a hardness of 119 mg/L (average hardness in Snap Lake in 2012), the site-specific benchmark for chloride in Snap Lake would be 353 mg/L. In 2012, chloride concentrations throughout Snap Lake were well below 353 mg/L.

Nitrate

Approximately 3% of the 2012 nitrate samples collected in Snap Lake were above the CCME WQG for nitrate of 2.93 mg-N/L, with the maximum concentration of 3.22 mg-N/L being measured at station SNAP03. However, average nitrate concentrations in the different lake areas ranged from 1.6 mg/L to 2.7 mg/L, and remained below the CCME WQG in 2012 (Table 3-13).

- The observed nitrate concentrations are also not expected to cause adverse effects to aquatic biota in Snap Lake because the toxicity of nitrate decreases with increases in hardness. BHP Billiton Canada Inc. (BHP Billiton) recently developed a hardness-dependent site-specific water quality objective for nitrate for the Ekati Diamond Mine. The objective was based on chronic toxicity test data that were available in the literature and from an investigation of the effect that water hardness has on the toxicity of nitrate (Elphick 2011; Rescan 2012; WLWB 2013). Using the formula provided in the BHP Billiton study and a hardness of 119 mg/L, a nitrate site-specific benchmark for Snap Lake was calculated to be 12 mg/L. In 2012, nitrate concentrations throughout Snap Lake were below 12 mg/L.

Fluoride

Similar to 2011 (De Beers 2012b), fluoride concentrations in the majority (i.e., 58%) of samples collected in 2012 were higher than the 2001 interim CCME (1999) WQG for inorganic fluorides of 0.12 mg/L. The maximum fluoride concentration measured in 2012 was 0.18 mg/L at station SNP 02-20d.

Based on the current conditions in Snap Lake, the observed fluoride concentrations are not expected to cause adverse effects to aquatic biota in Snap Lake because:

- The source of the elevated fluoride is seepage of shallow groundwater into the underground Mine (Drysdale 2011). Over time, this shallow groundwater is predicted to be replaced with lake water, which is lower in fluoride, thus concentrations are expected to decline (De Beers 2011b). Additionally, inflows to the Mine will be comprised of deeper groundwater, which is also lower in fluoride. Fluoride concentrations in Snap Lake will continue to be monitored to determine whether concentrations decrease over time as indicated by 2011 model predictions (De Beers 2011b).
- The CCME WQG for fluoride includes a relatively large safety factor (CCME 2002). It was derived from the lowest acceptable adverse effect level reported: a 144-hour lethal concentration (LC_{50}) value of 11.5 mg/L for the caddisfly *Hydropsyche bronta* (Environment Canada 2001). A safety factor of 100 was applied to this endpoint because it was an acute, lethal endpoint.
- Toxicity of fluoride is expected to decrease with increases in hardness, chloride, and calcium (Environment Canada 2001). Because fluoride, calcium, and chloride are constituents of the treated effluent, increases in calcium, chloride, and hardness are expected to always accompany increases in fluoride. In 2012, the average hardness concentration in Snap Lake was approximately 119 mg/L. The British Columbia Ministry of the Environment (BCMOE) recently published new guidance on calculating WQGs for fluoride using hardness (BCMOE 2011). Using the formula provided, and a hardness of 119 mg/L, the BCMOE recommended WQG is 1.4 mg/L, almost an order of magnitude higher than the maximum concentration measured in Snap Lake.

Proposed site-specific benchmarks and management actions for nitrate and TDS (which latter includes chloride and fluoride) are currently under development for the Mine as part of the Nitrogen and TDS Response Plans, respectively. In accordance with the Water Licence (MVLWB 2012), these plans will include a description of the sources of nitrogen and TDS, a description of the ecological implications of nitrogen and TDS loadings on the receiving environment, and a discussion on options for reducing loadings.

Dissolved Oxygen

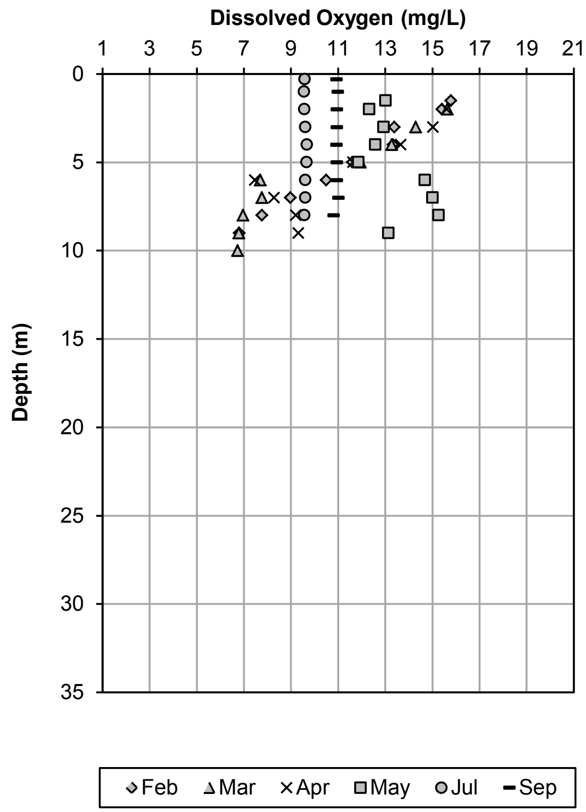
The DO concentration in lake water is a function of the balance of the processes that introduce or supplement oxygen into the water column and remove oxygen from the water column. Re-aeration by the diffuser and wind-mixing, as well as photosynthesis by algae and aquatic plants are prime examples of 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 2012, DO concentrations in Snap Lake were at levels considered healthy for fish and other aquatic organisms, with the exception of four locations (Figure 3-10) where field DO readings dropped below the CCME WQG of 6.5 milligrams per litre (mg/L) (CCME 1999). At these locations, the low DO only occurred in the bottom 0.5 m of the water column, indicating that the probe was likely near the sediment boundary or submerged in sediment as denoted by substantial changes in DO. As outlined in Appendix 3A, DO data from this area were only excluded from the assessment if there was a corresponding notable change in pH, temperature, and/or conductivity, indicating a change in substrate.

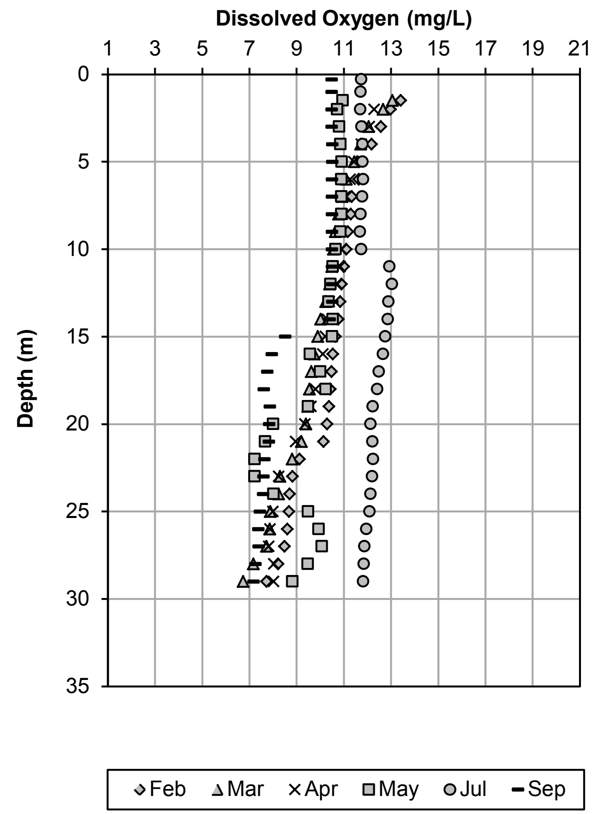
Since monitoring began in 2007, low DO concentrations near the bottom of the lake have been observed (De Beers 2002). Overall, DO concentrations in Snap Lake do not appear to have decreased as a result of treated effluent discharge. Increases, rather than decreases, have occurred over time as presented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

Figure 3-10 Dissolved Oxygen Concentrations in Snap Lake and Northeast Lake, 2012

a. Northwest Arm (SNAP02A)



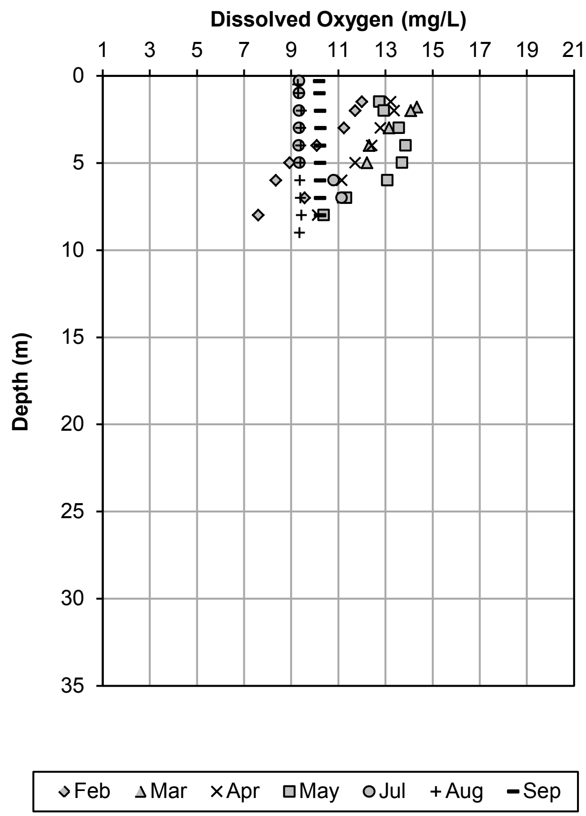
b. Northwest Arm (SNAP20B)



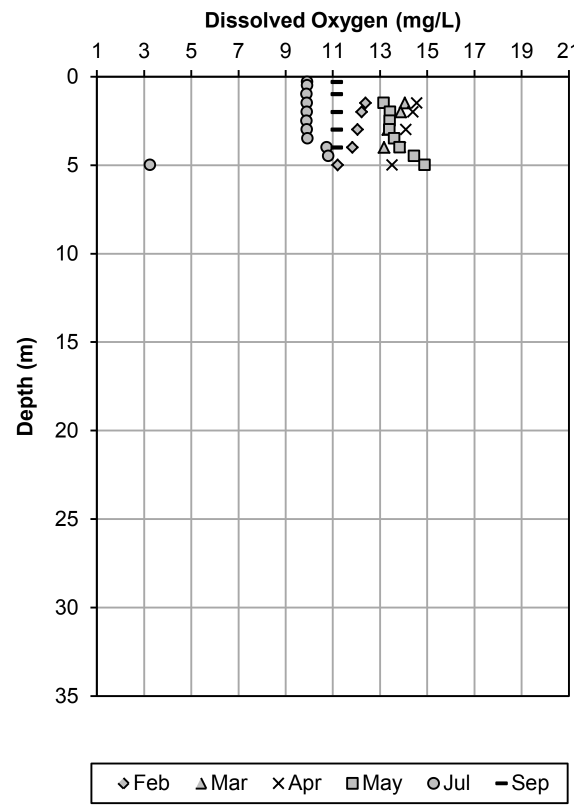
m = metres; mg/L = milligrams per litre.

Figure 3-10 Dissolved Oxygen Concentrations in Snap Lake and Northeast Lake, 2012

c. Far-field (SNAP08)

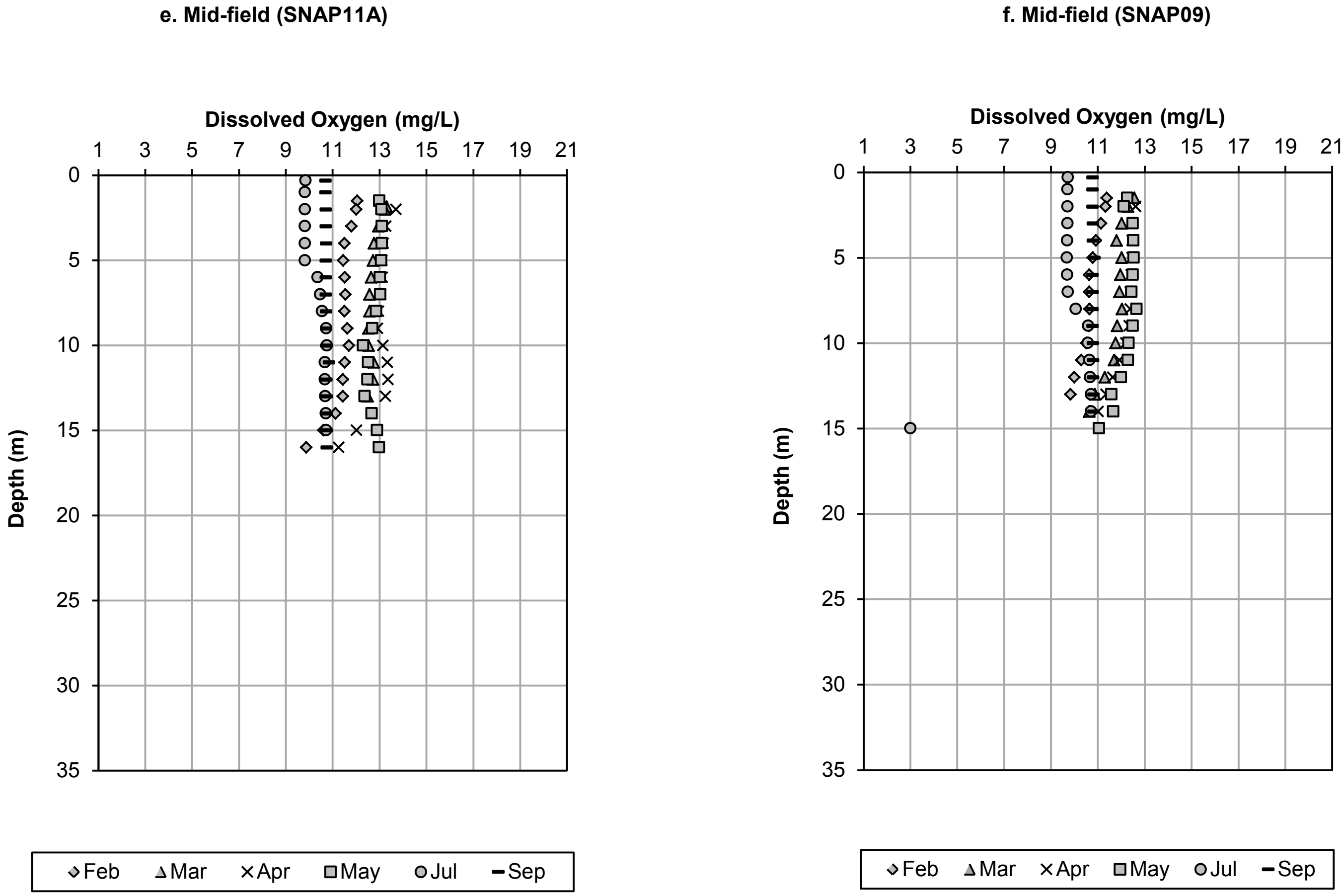


d. Far-field (SNAP10)



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

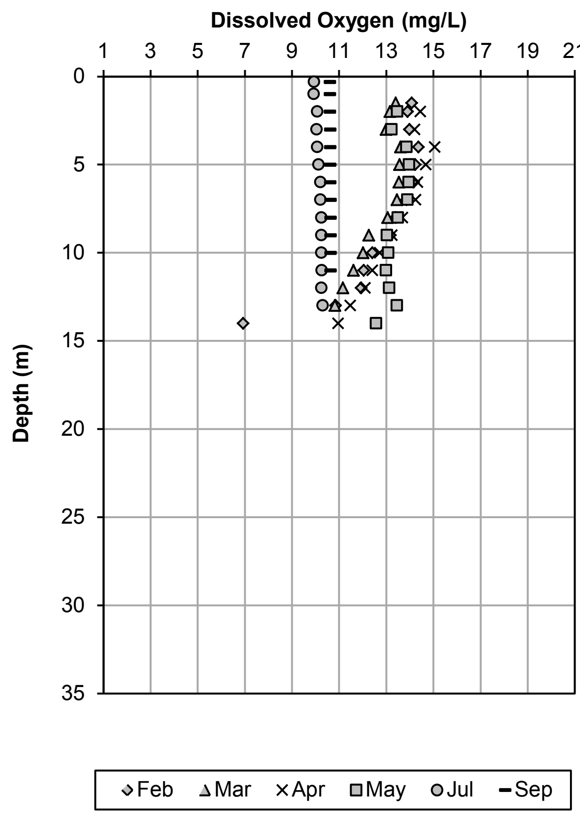
Figure 3-10 Dissolved Oxygen Concentrations in Snap Lake and Northeast Lake, 2012



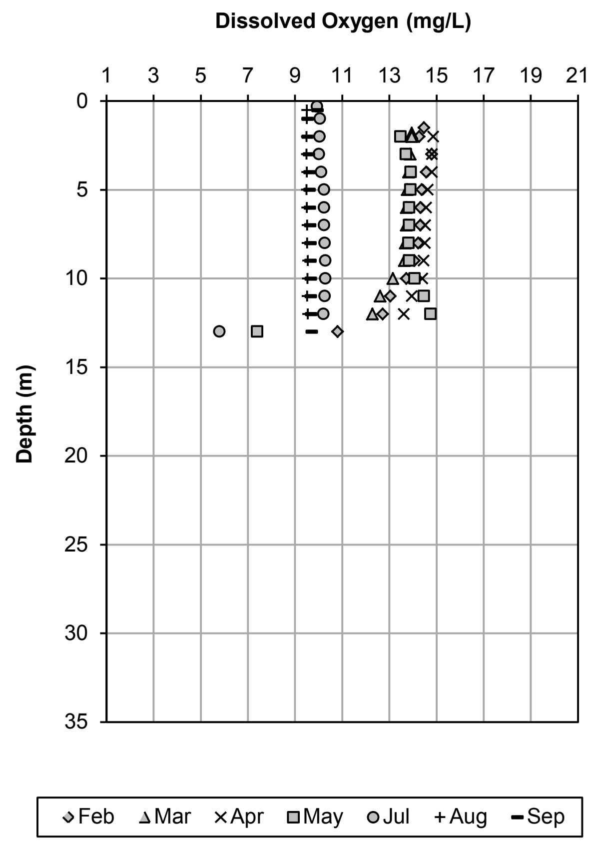
m = metres; mg/L = milligrams per litre.

Figure 3-10 Dissolved Oxygen Concentrations in Snap Lake and Northeast Lake, 2012

g. Near-field (SNAP05)



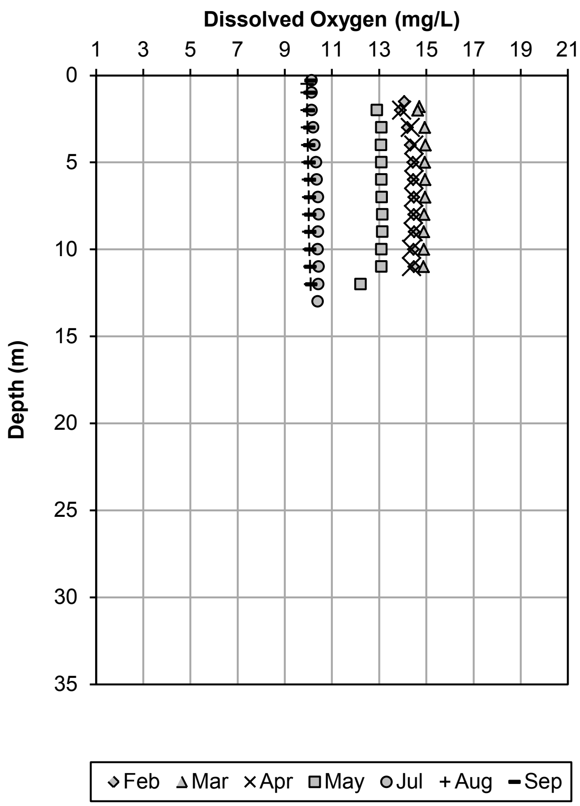
h. Near-field (SNAP03)



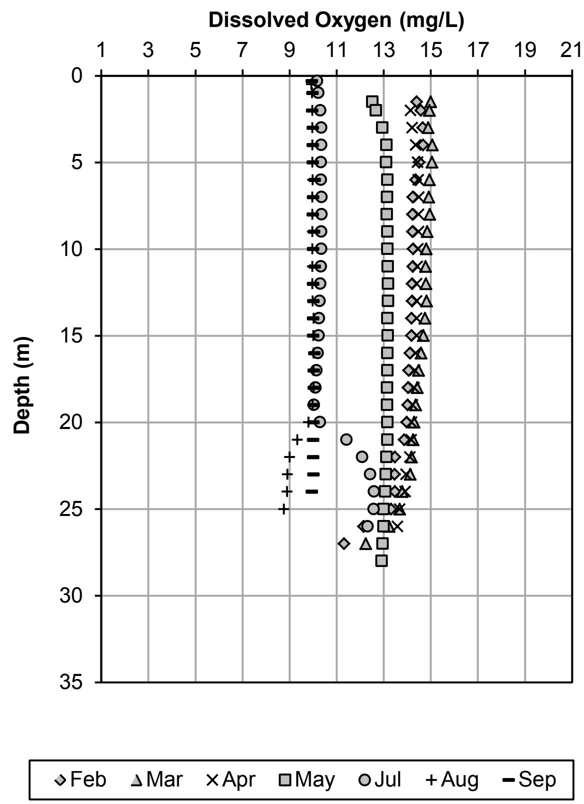
m = metres; mg/L = milligrams per litre.

Figure 3-10 Dissolved Oxygen Concentrations in Snap Lake and Northeast Lake, 2012

i. Diffuser (SNP02-20d)



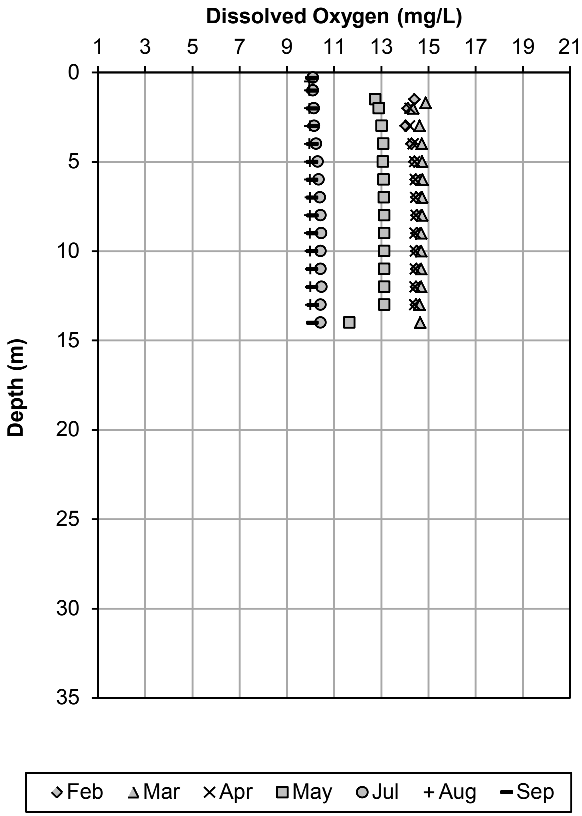
j. Diffuser (SNP02-20e)



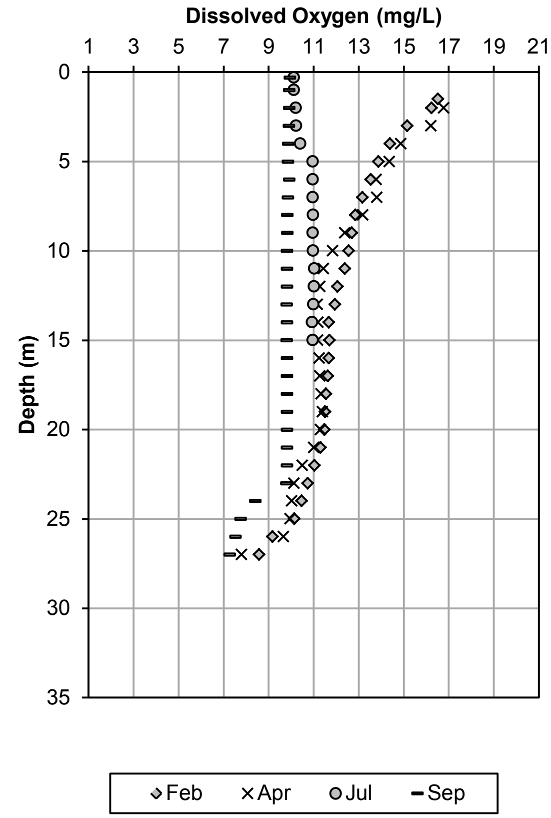
m = metres; mg/L = milligrams per litre.

Figure 3-10 Dissolved Oxygen Concentrations in Snap Lake and Northeast Lake, 2012

k. Diffuser (SNP02-20f)



l. Northeast Lake (NEL06)



Note:
Field DO profile in July 2012 at NEL06 was performed at a slightly different location; therefore, the deepest reading in July was at 15 m depth.
m = metres; mg/L = milligrams per litre.

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

Whole-lake average TDS concentrations in 2012 ranged from 187 to 234 mg/L and were below the Water Licence limit of 350 mg/L (Table 3-14). Temporal trends for TDS are provided in the response to Key Question 3 (Section 3.4.4).

The proposed TDS Response Plan will include detailed discussion of TDS sources and management as well as recommendations and supporting rationale for an appropriate site-specific water quality objective for TDS.

Table 3-14 Comparison of 2012 Water Quality to Water Licence Limit

Parameter	Units	Water Licence Limit[(a)]	Season			
			Ice-Covered		Open-Water	
			February	April	July	September
Total dissolved solids	mg/L	350	224	234	187	195

(a) Water Licence limit issued in MV2011L2-0004 (MVLWB 2012).
mg/L = milligram per litre.

3.4.3.3 Summary of Key Question 2

The 2012 water quality data from Snap Lake were below AEMP benchmarks and Water Licence limits, with the exception of chloride, fluoride, and nitrate. Similar to 2011, the majority of the 2012 fluoride concentrations were above the CCME WQG of 0.12 mg/L, which is a conservative concentration as it includes a relatively large safety factor (CCME 2002). Chloride concentrations in Snap Lake were typically below the CCME WQG of 120 mg/L for chloride with the exception of one chloride result of 121 mg/L collected from the diffuser area. Approximately 3% of the 2012 nitrate samples collected in Snap Lake were above the CCME WQG for nitrate of 2.93 mg-N/L, with a maximum concentration of 3.22 mg-N/L. Average nitrate concentrations in the different lake areas remained below the CCME WQG for nitrate. Whole-lake average concentration ranged from 187 mg/L to 234 mg/L, with a maximum TDS concentration of 279 mg/L, all below the Water License limit of 350 mg/L.

Because the primary source of fluoride, chloride, and nitrate is the treated effluent, increases in these parameters are associated with elevated calcium and hardness, which are expected to reduce the potential for toxicity effects associated with fluoride, chloride, and nitrate. Proposed site-specific benchmarks and management actions for TDS (which includes chloride and fluoride) and nitrate are currently under development for the Snap Lake Mine as part of the Nitrogen and TDS Response Plans, respectively.

In 2012, DO concentrations in Snap Lake were considered healthy for fish and other aquatic organisms, with the exception of four locations, where field DO readings dropped below the CCME WQG of 6.5 mg/L. At 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. Low DO concentrations near the bottom of the lake were observed during ice-covered conditions under baseline conditions. In 2012, increases rather than reductions in bottom DO concentrations were observed around the diffuser relative to Northeast Lake.

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?

3.4.4.1 Temporal Trends

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 2012 temporal assessment are described in the following subsections.

Screening and Visual Evaluation of Temporal Plots

Correlation analysis identified 42 water quality parameters with concentrations significantly related to conductivity in Snap Lake between 2004 and 2012 (Table 3-15). Parameters that were strongly correlated to conductivity were considered chemical signatures of treated effluent influence. Within this group of parameters, 22 increased in both the main basin and the northwest arm and 4 increased in only the main basin. Increasing concentrations of TDS, most major ions, most nitrogen parameters, and seven metals (i.e., barium, boron, lithium, molybdenum, nickel, rubidium, and strontium) were observed throughout Snap Lake. Fluoride, reactive silica, nitrite, and total uranium showed increasing trends in the main basin of Snap Lake (Table 3-15).

Increases in nitrate, total molybdenum, nickel, and rubidium in the northwest arm were new in 2012. An increasing trend in reactive silica was noted for the first time in 2012. The greater number of increasing trends currently identified in the northwest arm indicates that the effluent is gradually changing the quality of water within the northwest arm.

Temporal trends were not observed in Northeast Lake and Lake 13 between 2004 and 2012 for any of the measured parameters. Concentrations of parameters with increasing trends were above the Snap Lake normal range (i.e., baseline mean ± two standard deviations) and reference lake concentrations in at least one area of Snap Lake (Table 3-15). The lower concentrations and absence of trends in the Northeast Lake indicate that treated effluent exposure from the Mine is the primary contributor to the observed concentration increases for conductivity, TDS, major ions, nitrogen parameters, and eight metals in Snap Lake.

Of the parameters that increased, nitrate, chloride, and fluoride were above AEMP benchmarks (Section 3.4.3). Nitrite, ammonia, total boron, molybdenum, nickel, and uranium were all below

AEMP benchmarks. Most of the major ions (e.g., calcium, magnesium, potassium, sodium, sulphate) do not have AEMP benchmarks because CCME has not defined WQGs for these parameters. However, the major ions are implicit to TDS, so will be considered as part of the TDS Response Plan. Barium, lithium, rubidium, and strontium do not have AEMP benchmarks. A site-specific benchmark for strontium is being prepared; however, it is recommended that for the remaining parameters, total barium, lithium, and rubidium, available toxicological literature be reviewed to determine the implications of these parameters increasing beyond the normal range.

Fifteen parameters had weak (or low) correlations with conductivity (i.e., turbidity, dissolved inorganic phosphorus, dissolved phosphorus, total phosphorus, total inorganic phosphorus, total arsenic, chromium, copper, iron, lead, mercury, manganese, titanium, vanadium and zinc; Table 3-15). Temporal trends were not observed for these parameters in the main basin or northwest arm of Snap Lake. Parameters such as total organic carbon, total antimony, cadmium, silver, zinc, and methyl mercury had no correlation to conductivity.

Examples of increasing trends are shown for stations representative of different areas within Snap Lake (i.e., diffuser, far-field, northwest arm) for TDS, nitrate, ammonia, and strontium (Figures 3-11 to 3-14, respectively). Additional temporal plots of parameters from all five areas in Snap Lake and from Northeast Lake and Lake 13 are presented in Appendix A6, Figures A6-2 to A6-51.

Table 3-15 Summaries of Temporal Trends for Parameters That Were Significantly Correlated With Laboratory Conductivity

Parameter	Pearson Correlation Coefficient	Strength of Correlation Coefficient[a]	Temporal Trends in Snap Lake Area (2004 to 2012)					2012 Concentration Above Snap Lake Normal Range?	2012 Concentration Above Reference Lakes?
			Diffuser	Near-field	Mid-field	Far-field	Northwest Arm		
Conventional Parameters									
Laboratory pH	0.645	Moderate	↑	↑	↑	↑	↑	yes	no
Total Dissolved Solids, Calculated (Standard Methods)	0.998	High	↑	↑	↑	↑	↑	yes	yes
Turbidity	-0.175	Low	-	-	-	-	-	no	no
Ions									
Bicarbonate, as HCO_3	0.905	High	↑	↑	↑	↑	↑	yes[b]	yes[b]
Calcium	0.995	High	↑	↑	↑	↑	↑	yes	yes
Chloride	0.996	High	↑	↑	↑	↑	↑	yes	yes
Fluoride	0.854	High	↑	↑	↑	↑	-	yes[c]	yes[c]
Hardness, as $CaCO_3$	0.995	High	↑	↑	↑	↑	↑	yes	yes
Magnesium	0.977	High	↑	↑	↑	↑	↑	yes	yes
Potassium	0.943	High	↑	↑	↑	↑	↑	yes	yes[c]
Reactive Silica, as SiO_2	0.699	Moderate	↑	↑	↑	↑	-	yes[c]	yes[c]
Sodium	0.994	High	↑	↑	↑	↑	↑	yes	yes
Sulphate	0.985	High	↑	↑	↑	↑	↑	yes	yes
Total Alkalinity, as $CaCO_3$	0.904	High	↑	↑	↑	↑	↑	yes	yes
Nutrients									
Dissolved Inorganic Phosphorus	-0.17	Low	-	-	-	-	-	no	no
Dissolved Phosphorus	-0.215	Low	-	-	-	-	-	no	no
Nitrate, as N, Calculated	0.927	High	↑	↑	↑	↑	↑	yes	no
Nitrate/Nitrite, as N	0.924	High	↑	↑	↑	↑	↑	yes	yes
Nitrite, as N	0.851	High	↑	↑	↑	↑	-	yes	yes[c]
Total Ammonia, as N	0.787	High	↑	↑	↑	↑	↑	yes[b]	yes[b]
Total Inorganic Phosphorus	-0.221	Low	-	-	-	-	-	no	no
Total Kjeldahl Nitrogen	0.623	Moderate	↑	↑	↑	↑	↑	no	yes[c]
Total Phosphorus	-0.253	Low	-	-	-	-	-	no	no
Total Metals									
Total Aluminum	-0.306	Moderate	-	-	-	-	-	no	no
Total Arsenic	0.113	Low	-	-	-	-	-	no	no
Total Barium	0.965	High	↑	↑	↑	↑	↑	yes	yes
Total Boron	0.956	High	↑	↑	↑	↑	↑	yes	yes
Total Chromium	-0.205	Low	-	-	-	-	-	no	no
Total Copper	-0.181	Low	-	-	-	-	-	no	no
Total Iron	-0.126	Low	-	-	-	-	-	no	no

Table 3-15 Summaries of Temporal Trends for Parameters That Were Significantly Correlated With Laboratory Conductivity

Parameter	Pearson Correlation Coefficient	Strength of Correlation Coefficient[(a)]	Temporal Trends in Snap Lake Area (2004 to 2012)					2012 Concentration Above Snap Lake Normal Range?	2012 Concentration Above Reference Lakes?
			Diffuser	Near-field	Mid-field	Far-field	Northwest Arm		
Total Lead	-0.219	Low	-	-	-	-	-	no	no
Total Lithium	0.983	High	↑	↑	↑	↑	↑	yes[(b)]	yes[(b)]
Total Manganese	0.106	Low	-	-	-	-	-	yes[(d)]	yes[(d)]
Total Mercury	-0.189	Low	-	-	-	-	-	no	no
Total Mercury (Flett)	-0.352	Moderate	-	-	-	-	-	no	no
Total Molybdenum	0.939	High	↑	↑	↑	↑	↑	yes	yes
Total Nickel	0.833	High	↑	↑	↑	↑	↑	yes[(e)]	yes
Total Rubidium	0.897	High	↑	↑	↑	↑	↑	yes	yes
Total Strontium	0.992	High	↑	↑	↑	↑	↑	yes	yes
Total Thallium	-0.306	Moderate	-	-	-	-	-	no	no
Total Titanium	0.209	Low	-	-	-	-	-	no	no
Total Uranium	0.848	High	↑	↑	↑	↑	-	no	no
Total Vanadium	0.046	Low	-	-	-	-	-	no	no
Total Zinc	-0.044	Low	-	-	-	-	-	no	no

Note: ↑ = an increasing trend; - = indicates no relative increasing or decreasing trend; normal range is based on data collected prior to 2004, with the upper and lower range calculated as the mean ± 2 standard deviations. Normal range is based on data collected prior to 2004, with the upper and lower range calculated as the mean ± 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 main basin; above during ice-cover in northwest arm.

(c) Above in main basin only.

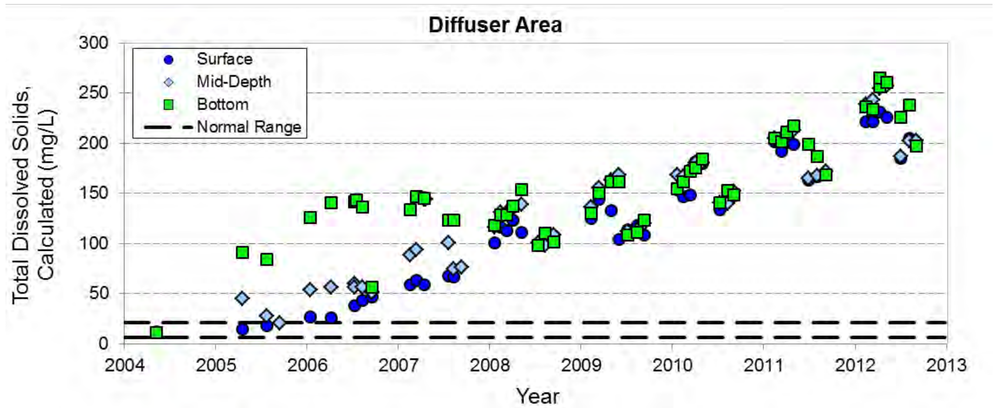
(d) Above in diffuser area.

(e) Near-field (open-water), mid-field (open-water), far-field and northwest arm within normal range.

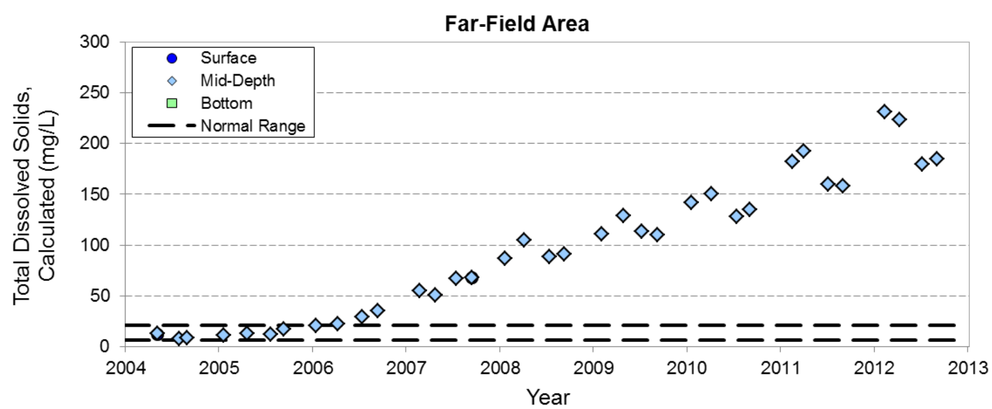
$CaCO_3$ = calcium carbonate; HCO_3 = bicarbonate; N = nitrogen; P = phosphorus; SiO_2 = silicate

Figure 3-11 Total Dissolved Solids Concentrations in Snap Lake, 2004 to 2012

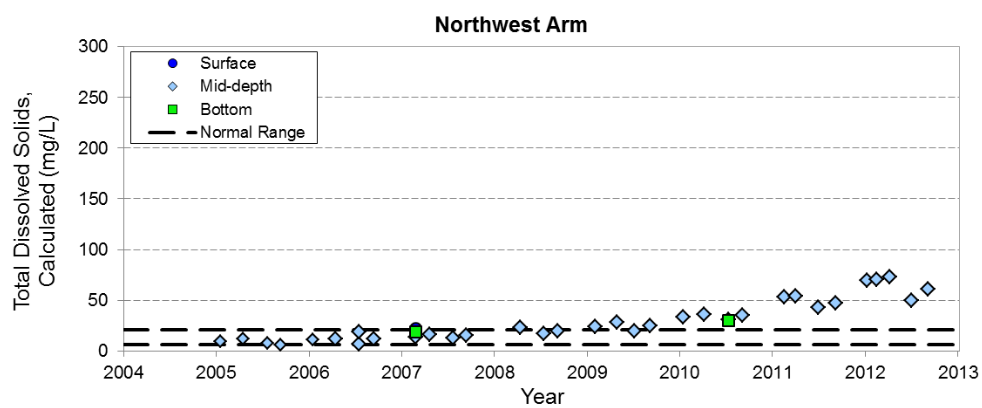
a. Diffuser Area



b. Far-Field Area



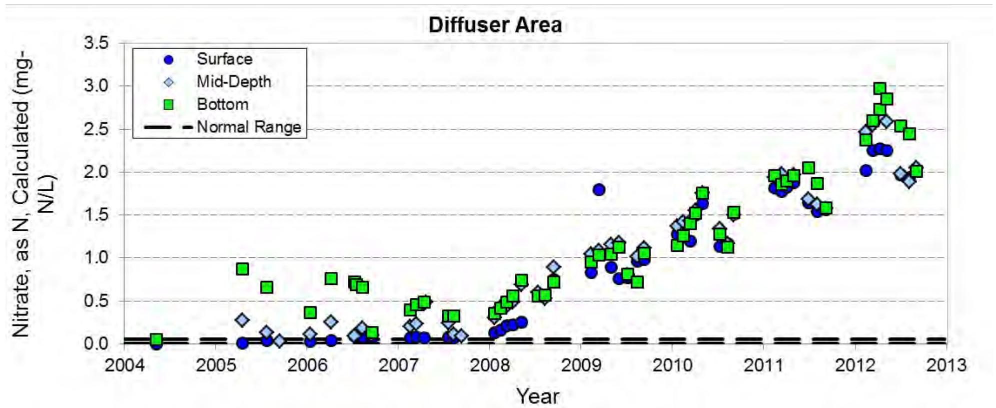
b. Northwest Arm



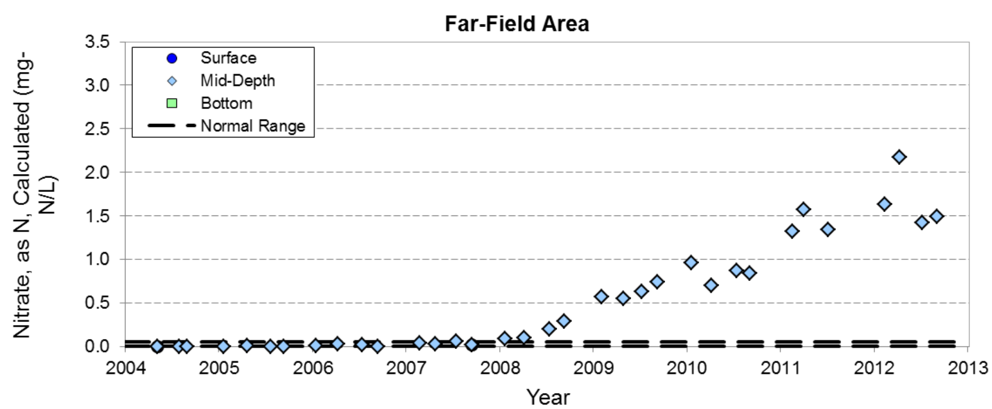
Note: Normal range is based on data collected prior to 2004, with the upper and lower range calculated as the mean ± 2 standard deviations; data shown are from representative stations within Snap Lake: Diffuser Area = SNAP13 (2004 to April 2006) and SNP 02-20e (July 2006 to 2012); Far-field Area = SNAP08; Northwest Arm = SNAP02 (2004 to April 2006) and SNAP02A (July 2006 to 2012).
mg/L = milligrams per litre.

Figure 3-12 Nitrate Concentrations in Snap Lake, 2004 to 2012

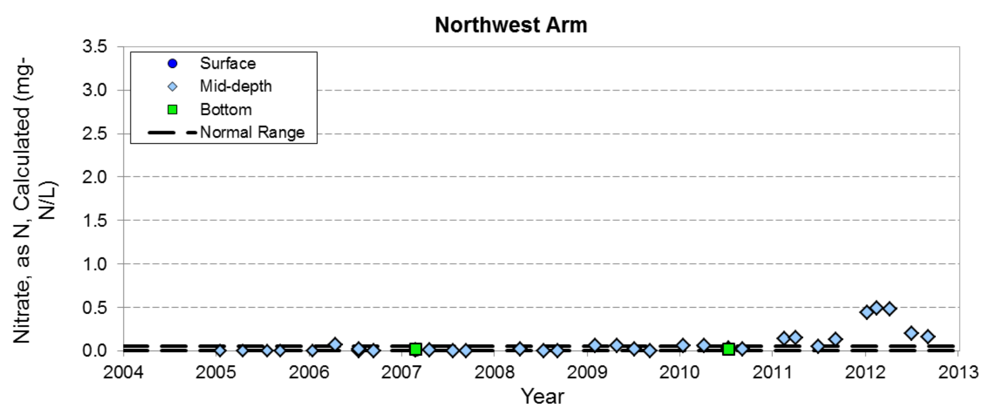
a. Diffuser Area



b. Far-Field Area



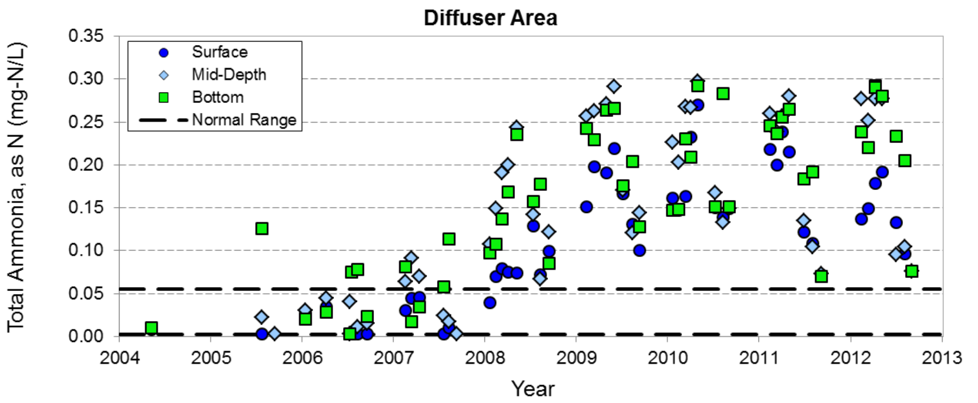
c. Northwest Arm



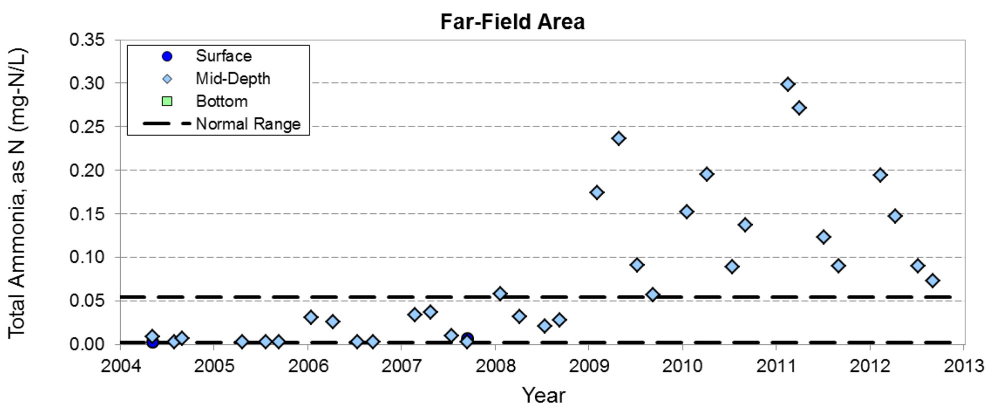
Note: Normal range is based on data collected prior to 2004, with the upper and lower range calculated as the mean ± 2 standard deviations; data shown are from representative stations within Snap Lake: Diffuser Area = SNAP13 (2004 to April 2006) and SNP 02-20e (July 2006 to 2012); Far-field Area = SNAP08; Northwest Arm = SNAP02 (2004 to April 2006) and SNAP02A (July 2006 to 2012).
mg-N/L = milligrams as nitrogen per litre.

Figure 3-13 Ammonia Concentrations in Snap Lake, 2004 to 2012

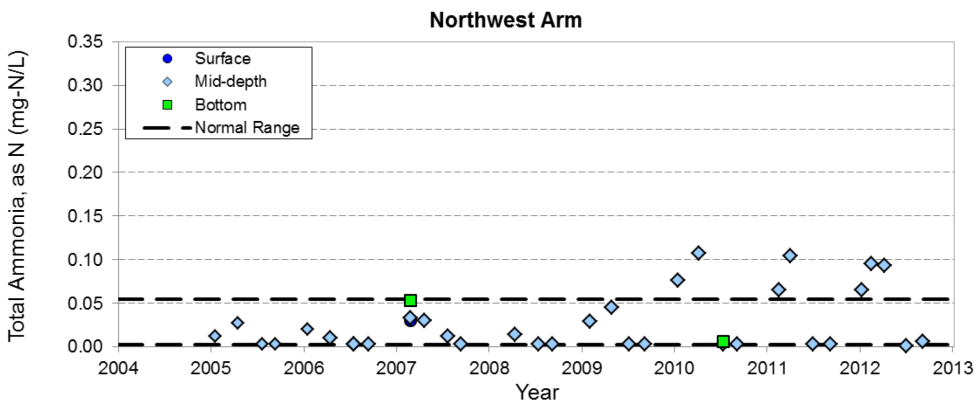
a. Diffuser Area



b. Far-Field Area



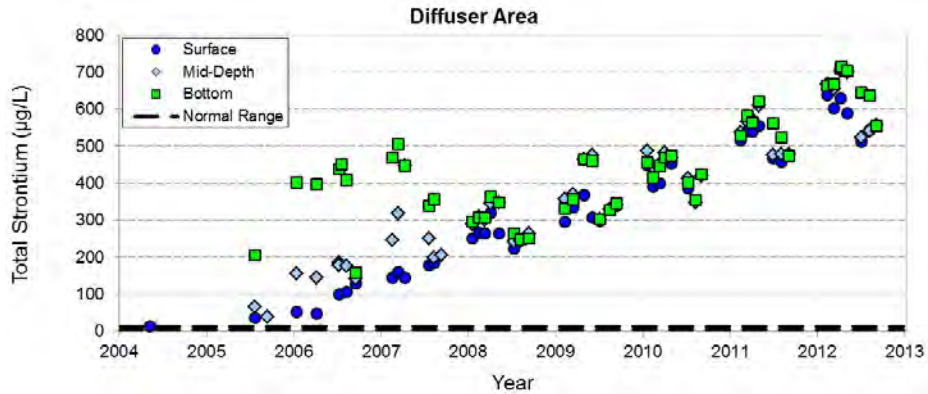
c. Northwest Arm



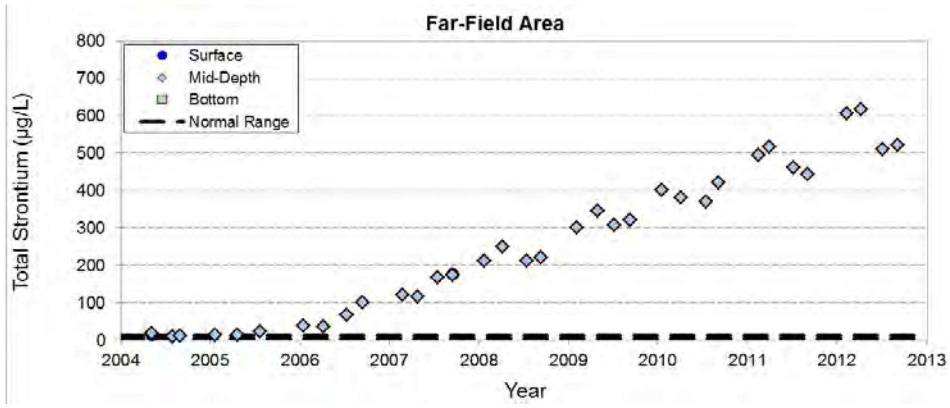
Note: Normal range is based on data collected prior to 2004, with the upper and lower range calculated as the mean ± 2 standard deviations; data shown are from representative stations within Snap Lake: Diffuser Area = SNAP13 (2004 to April 2006) and SNP 02-20e (July 2006 to 2012); Far-field Area = SNAP08; Northwest Arm = SNAP02 (2004 to April 2006) and SNAP02A (July 2006 to 2012).
mg-N/L = milligrams as nitrogen per litre.

Figure 3-14 Total Strontium Concentrations in Snap Lake, 2004 to 2012

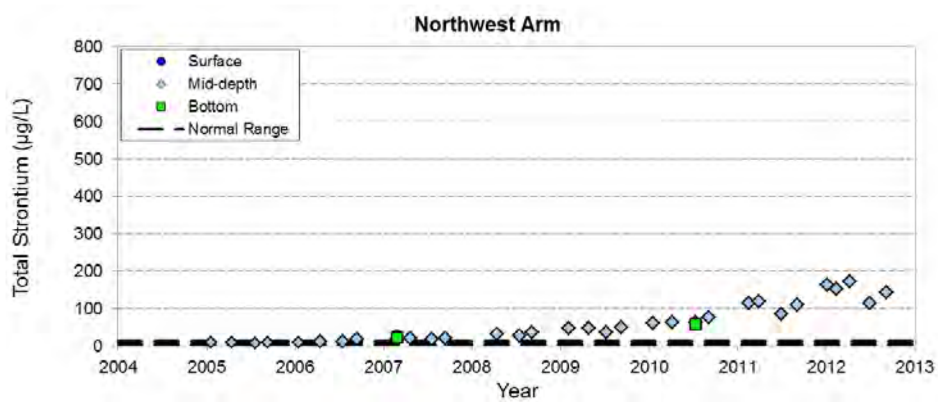
a. Diffuser Area



b. Far-Field Area



c. Northwest Arm



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: Diffuser Area = SNAP 13 (2004 to April 2006) and SNP 02-20e (July 2006 to 2012); Far-field Area = SNAP08; Northwest Arm = SNAP02 (2004 to April 2006) and SNAP02A (July 2006 to 2012).
µg/L = micrograms per litre.

Comparison to EAR Predictions and 2011 Water Licence Renewal Application Model Predictions

Parameters Above AEMP Benchmarks

Increasing trends in TDS concentrations were observed in every area of Snap Lake, including the near-field area, far-field area close to the lake outlet, and the northwest arm (Figure 3-11). The greatest increase in TDS was observed in the diffuser area, where concentrations were above 260 mg/L. The smallest increase was observed in the northwest arm (Figure 3-11, panel c), where the maximum concentration at the representative station (SNAP02A) was approximately 80 mg/L. 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 3-11), confirming an increasing trend of treated effluent exposure in this area.

Concentrations of TDS in 2012 at the diffuser, mid-field, and far-field areas in Snap Lake were overlain on the EAR predictions in Figure 3-15. TDS concentrations within 200 m of the treated effluent discharge (i.e., diffuser area) were between EAR model predictions and 2011 Water Licence Renewal Application model predictions. There was a divergence between measured TDS concentrations and 2011 model predictions at the diffuser, mid-field, and far-field areas (Figure 3-15). The whole-lake average measured TDS concentrations and whole-lake average model predictions in 2011 and 2012 were also different (Figure 3-16). The divergence was due to higher than predicted TDS loadings from the treated effluent discharge in 2011 and 2012 (Figure 3-17). The Snap Lake water quality model (De Beers 2011b), developed for the 2011 Water Licence Renewal Application, which includes groundwater, site and lake components, is currently being updated to reflect these changes in loadings. Measured TDS concentrations from 2006 to 2012 at the mid-field and far-field areas in Snap Lake were greater than EAR model predictions. The difference between measured and modelled TDS concentrations in the mid-field and far-field areas may be due to an underestimation of circulation patterns and mixing within Snap Lake during the EAR modelling. Concentrations of TDS have increased throughout the lake faster than expected.

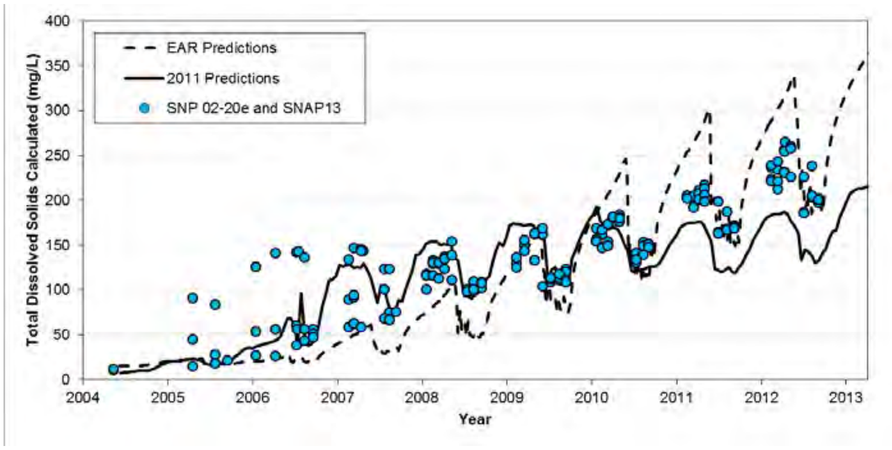
Nutrient concentrations were expected to increase over time in areas influenced by the discharge because the treated effluent contains elevated concentrations of both nitrogen and phosphorus. In particular, the treated effluent contains elevated concentrations of dissolved inorganic nitrogen, including nitrate and ammonia, from nitrogen-based explosives used in the mining process, and the treated domestic waste water, which contains nitrogen and phosphorus. Nitrate concentrations have increased since 2004 during both ice-covered and open-water conditions in the near-field and far-field areas (Figures 3-18 and 3-19). Nitrate in the northwest arm was also elevated during ice-covered conditions in 2012, compared to previous years.

Measured and predicted nitrate concentrations at the diffuser, mid-field, and far-field areas in Snap Lake are presented in Figure 3-18. 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, maximum measured nitrate concentrations in Snap Lake have remained below EAR model predictions. There is a divergence between measured nitrate concentrations in 2012 and 2011 model predictions at the diffuser, mid-field, and far-field areas (Figure 3-18), and between whole-lake average measured nitrate concentrations and 2011 whole-lake average model predictions in 2012 (Figure 3-19). Similar to TDS concentrations, the divergence in measured and modelled nitrate concentrations is due to higher than predicted nitrate loadings from the treated effluent discharge in 2011 and 2012.

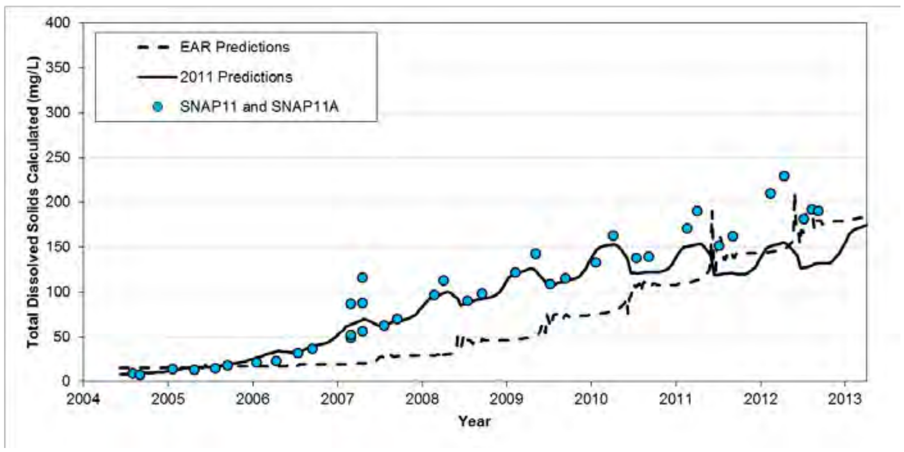
Forty seven out of 136 fluoride results from Snap Lake were above the maximum whole-lake average 2011 model prediction of 0.14 mg/L (De Beers 2011b). These samples were collected from the diffuser, near-field, mid-field, and far-field areas in Snap Lake. The maximum fluoride concentration measured in 2012 was 0.18 mg/L (Table 3-13). As described in Section 3.4.3, fluoride concentrations in Snap Lake are expected to decrease over time.

Figure 3-15 Measured and Predicted Total Dissolved Solids Concentrations in Snap Lake

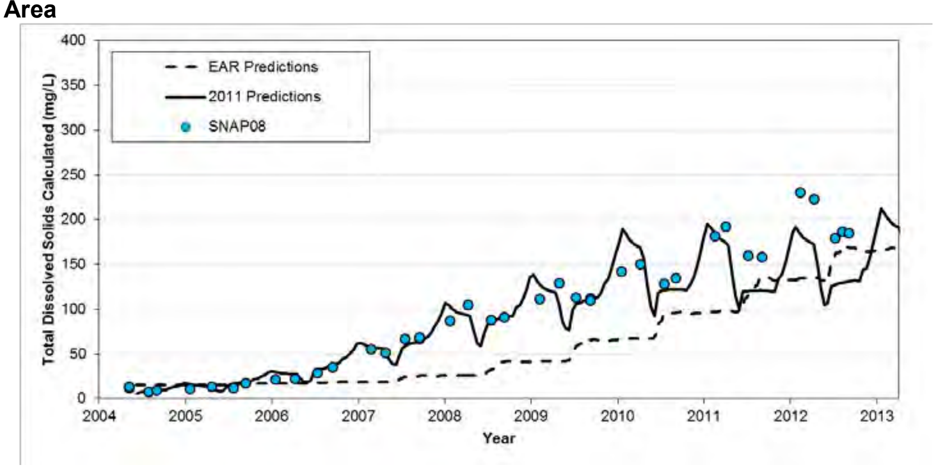
a. Diffuser Area



b. Mid-field Area

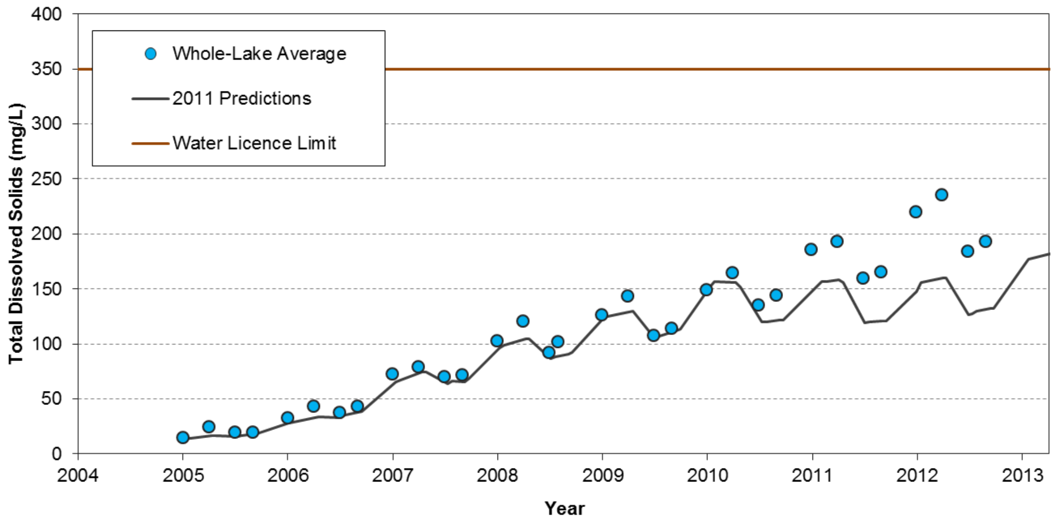


c. Far-field Area



Note: Data shown are from representative stations within Snap Lake: Diffuser Area = SNP 02-20e and SNAP13; Mid-field Area = SNAP11 and SNAP11A; Far-field Area = SNAP08; 2011 predictions represent the upper bound scenario (De Beers 2011b); EAR predictions are from De Beers (2002).
mg/L = milligrams per litre; m = metre.

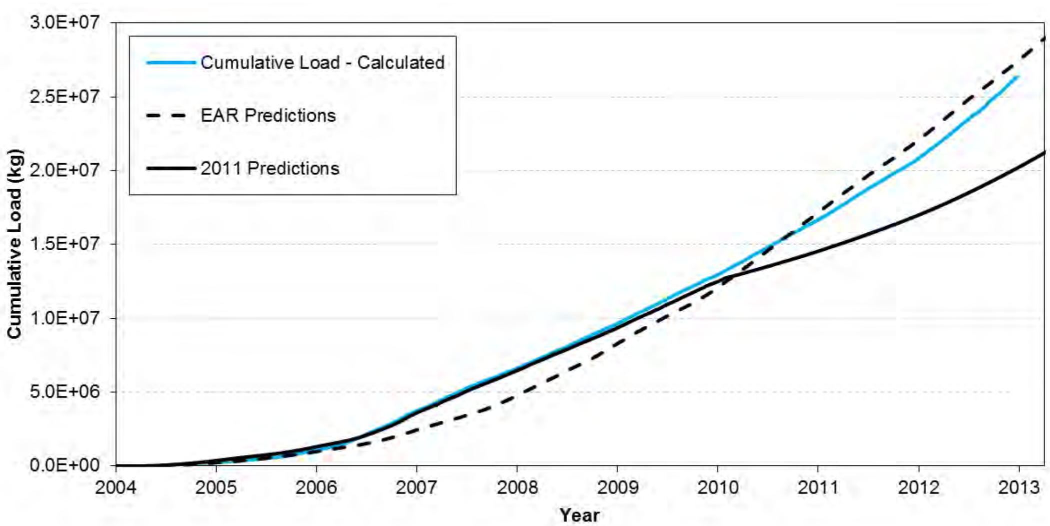
Figure 3-16 Measured and Predicted Whole-Lake Average Total Dissolved Solids Concentrations in Snap Lake



Note: 2011 prediction represents the upper bound scenario (De Beers 2011b); Water Licence limit issued in MV2011L2-0004 (MVLWB 2012).

mg/L = milligrams per litre.

Figure 3-17 Measured and Predicted Cumulative Load of Total Dissolved Solids in Treated Effluent

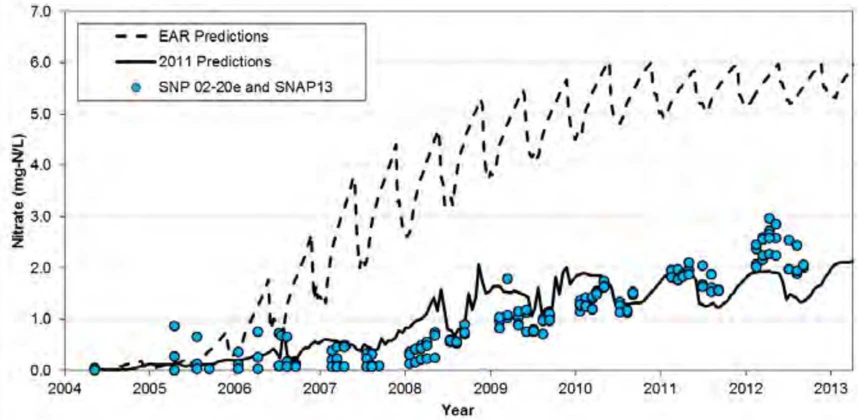


Note: 2011 predictions represent the upper bound scenario (De Beers 2011b); EAR predictions are from De Beers (2002).

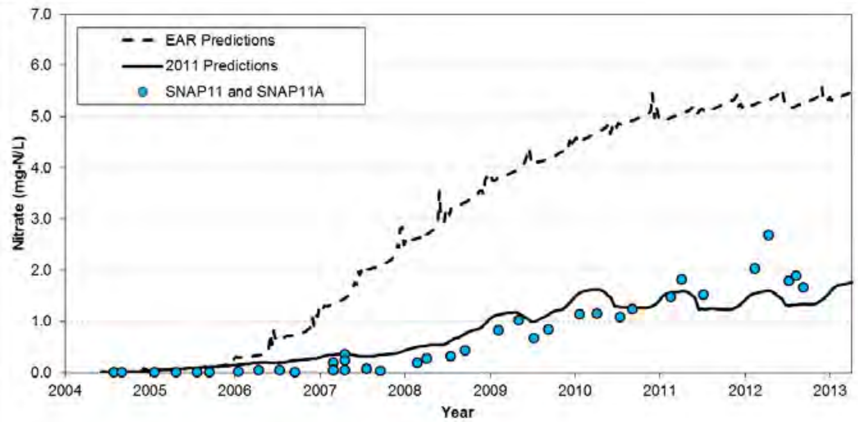
kg = kilograms.

Figure 3-18 Measured and Predicted Nitrate (as N) Concentrations in Snap Lake

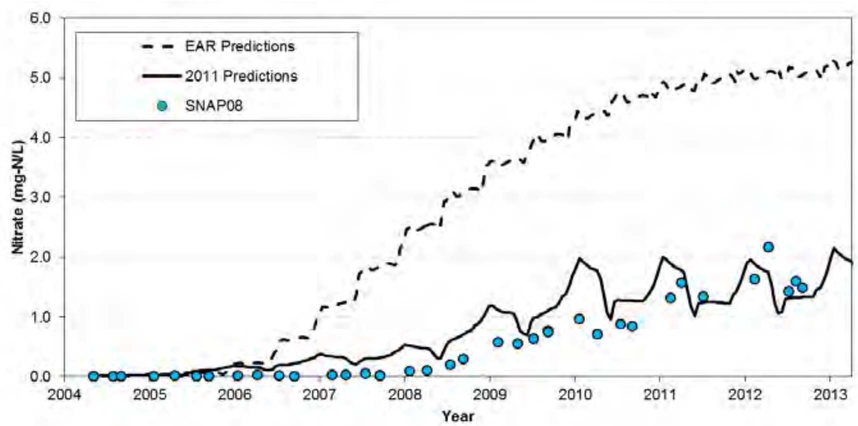
a. Diffuser Area



b. Mid-field Area

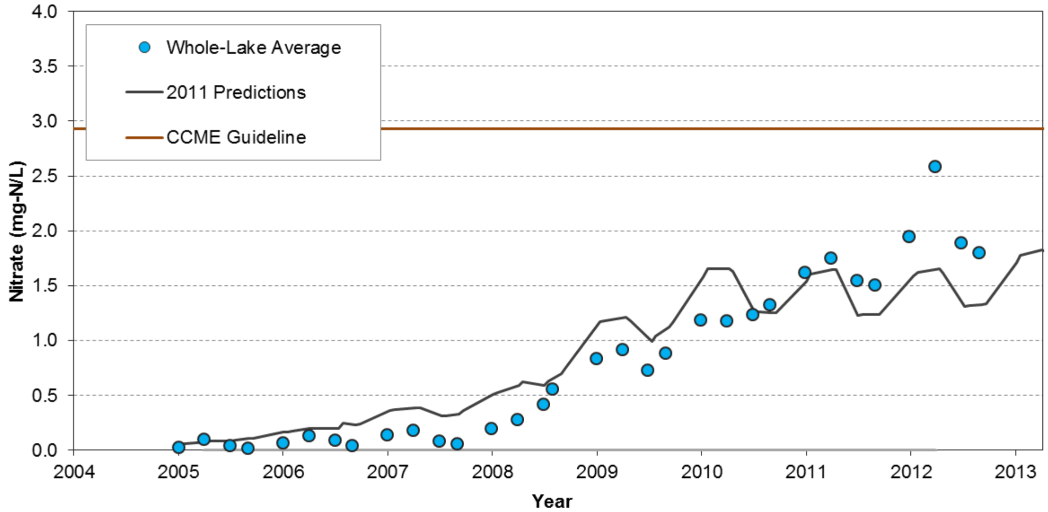


c. Far-field Area



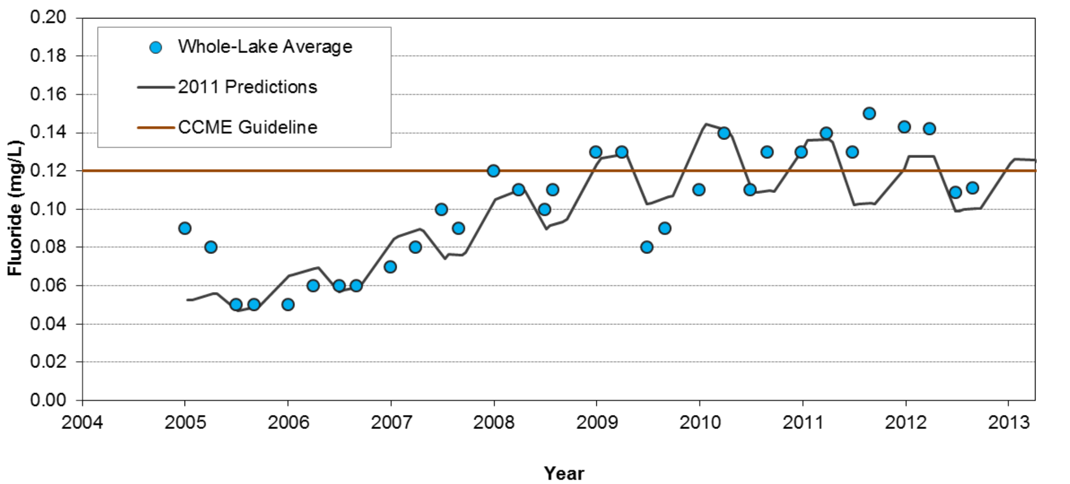
Note: Data shown are from representative stations within Snap Lake: Diffuser Area = SNP 02-20e and SNAP13; Mid-field Area = SNAP 11 and SNAP 11A; Far-field Area = SNAP08; 2011 predictions represent the upper bound scenario (De Beers 2011b); EAR predictions are from De Beers (2003).
mg-N/L = milligrams as nitrogen per litre.

Figure 3-19 Observed and Predicted Whole-Lake Average Nitrate (as N) Concentrations in Snap Lake



Note: 2011 prediction represents the upper bound scenario (De Beers 2011b); CCME guideline for nitrate is from CCME (1999).
mg-N/L = milligrams as nitrogen per litre.

Figure 3-20 Observed and Predicted Whole-Lake Average Fluoride Concentrations in Snap Lake



Note: 2011 prediction represents the upper bound scenario (De Beers 2011b); CCME guideline for fluoride is from CCME (1999).
mg/L = milligrams per litre.

Other Parameters

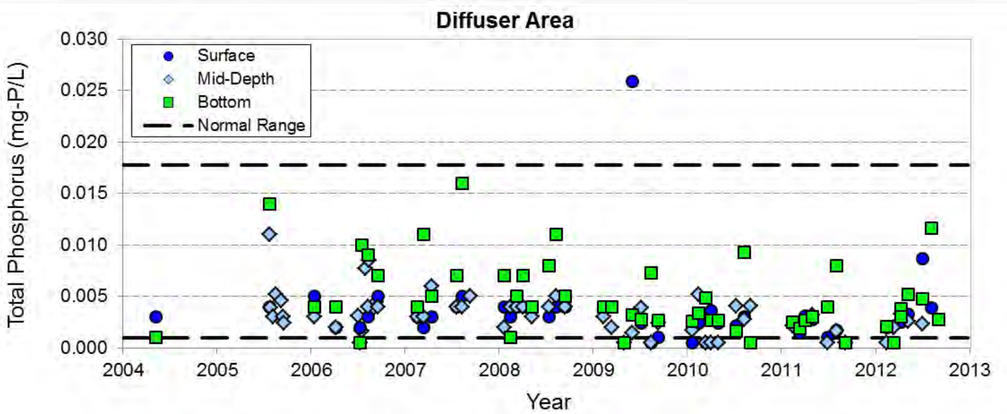
In contrast to nitrogen nutrients, there have been no obvious increases in TP concentrations in Snap Lake since 2004 (Figure 3-21). Between 2009 and 2012, water samples were collected during the phytoplankton and water quality programs of the AEMP, and then sent to different laboratories for analyses. A review of the data from the two simultaneous studies indicated that the results from the two laboratories were different. Increases in TP concentrations were not observed in samples collected from Snap Lake from 2004 to 2012 (Figure 3-21) even through there was a phosphorus load to Snap Lake from the treated effluent (Section 3.4.2). Possible explanations for the absence of a corresponding increase in TP concentrations include the following:

- Aquatic organisms may be rapidly taking up phosphorus released at the diffuser, and then dying off and settling to the bottom (Section 4).
- Phosphorus may be precipitating out of the water column and settling out in the bottom sediments near the diffuser.
- The littoral zone may be intercepting the phosphorus before it can be measured in the water column during open-water conditions (Section 12.1).
- Phosphorus concentrations in Snap Lake are near detection limits and, at these low concentrations, there is a greater degree of uncertainty in the laboratory reported concentrations, as described in the Nutrient Special Study (Section 12.4).
- Sampling depth may be a contributing factor; TP concentrations were higher in the euphotic zone samples compared to the corresponding mid-depth samples in 2012 (Section 12.4).

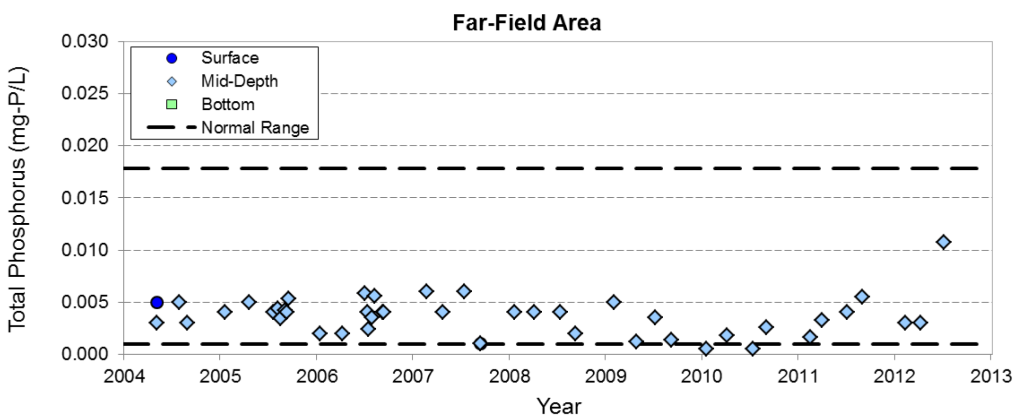
Although small changes in phosphorus concentrations have the potential to influence phytoplankton biomass and benthic invertebrate communities, such slight temporal trends may be difficult to detect in Snap Lake. The inherent uncertainty in the low-level phosphorus values in Snap Lake has the potential to lower the ability to detect slight temporal trends in phosphorus. To investigate this difference, and the uncertainty associated with low-level phosphorus measurements, a Nutrient Special Study was completed in 2012. Detailed methods, results and recommendations from this study are presented in Section 12.4 of this report.

Figure 3-21 Total Phosphorus Concentrations in Snap Lake, 2004 to 2012

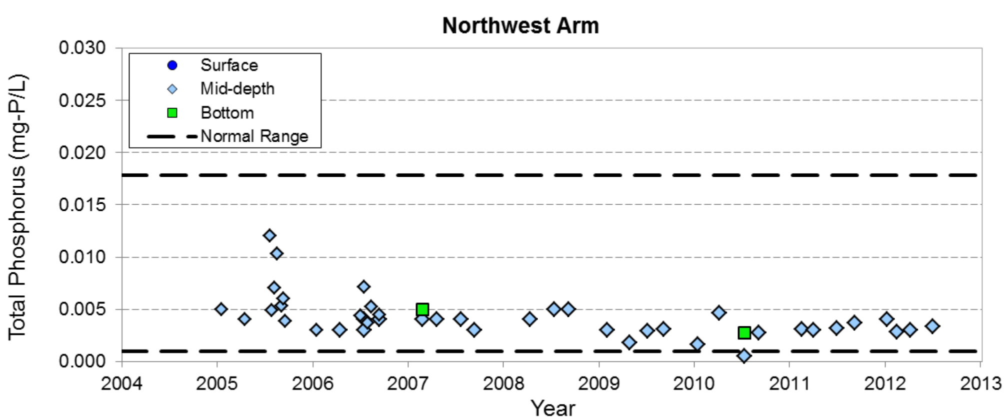
a. Diffuser Area



b. Far-Field Area



c. Northwest Arm



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: Diffuser Area = SNAP13 (2004 to April 2006) and SNP 02-20e (July 2006 to 2012); Far-field Area = SNAP08; Northwest Arm = SNAP02 (2004 to April 2006) and SNAP02A (July 2006 to 2012).
mg-P/L = milligrams as phosphorus per litre.

As an additional screening step, maximum concentrations of parameters measured in Snap Lake in 2012 were compared to maximum whole-lake annual average concentrations from the EAR to identify potential unexpected increases. In 2012, maximum concentrations for all laboratory parameters in Snap Lake were below the maximum whole-lake annual average EAR predictions with the exception of total cadmium, hexavalent chromium, and total manganese (Table 3-16). These exceptions are discussed in more detail below.

Table 3-16 Comparison of 2012 Snap Lake Water Quality to EAR Predictions and 2011 Water Licence Renewal Application Model Predictions

Parameter	Units	AEMP Benchmarks (Protection of Aquatic Life)[a]	EAR Predictions[b]	2011 Predictions[c]	Observed Concentrations[d]	
					Type	Snap Lake
Conventional Parameters						
Total dissolved solids, calculated (Standard Methods)	mg/L	-	350	558	max	279
Ions						
Chloride	mg/L	120	137	304	max	**121**
Fluoride	mg/L	0.12	-	0.14	max and range in average[e]	**0.18** and 0.05 to **0.13**
Sodium	mg/L	-	-	-	max	31
Calcium	mg/L	-	88	137	max	62
Magnesium	mg/L	-	9	5	max	8
Sulphate	mg/L	-	-	-	max	24
Nutrients						
Nitrate, as N	mg-N/L	2.93	5.87 / 6.00	3.62	max	**3.22**
Nitrite, as N	mg-N/L	-	-	-	max	0.029
Ammonia, as N	mg-N/L	1.0 to 21.6[f]	1.23 / 1.10	0.80	max	0.32
Total phosphorus	mg-P/L	-	0.013	-	range in average[e]	0.002 to 0.004
Total Metals						
Aluminum	µg/L	100[g]	-	15	max	15
Arsenic	µg/L	-	-	-	max	0.2
Barium	µg/L	-	-	-	max	29
Boron	µg/L	-	-	-	max	53
Cadmium	µg/L	0.36	0.058	0.039	max and range in average[e]	0.07 and 0.004 and 0.018
Chromium	µg/L	8.9	-	2.5	Max	0.3
Hexavalent chromium	µg/L	2.1	0.8	-	max and range in average[e]	1.2 and 0.01 to 0.09
Copper	µg/L	7.9	2.2	2.2	max	0.8
Iron	µg/L	-	-	-	Max	19
Lead	µg/L	1 to 7[h]	0.58	0.27	Max	0.07
Lithium	µg/L	-	-	-	Max	11
Manganese	µg/L	-	19	-	max and range in average[e]	39 and 2 to 7
Mercury (Flett)	µg/L	0.026	-	0.029	max	0.0012
Molybdenum	µg/L	73	-	3	max	1.5
Nickel	µg/L	33.9 to 153[h]	8.1	4.1	max	2.2
Selenium	µg/L	1	0.42	-	max	0.044
Silver	µg/L	0.1	-	-	max	<0.005
Thallium	µg/L	0.8	-	-	max	<0.01

Table 3-16 Comparison of 2012 Snap Lake Water Quality to EAR Predictions and 2011 Water Licence Renewal Application Model Predictions

Parameter	Units	AEMP Benchmarks (Protection of Aquatic Life)[(a)]	EAR Predictions[(b)]	2011 Predictions[(c)]	Observed Concentrations[(d)]	
					Type	Snap Lake
Uranium	µg/L	15	-	-	max	0.2
Zinc	µg/L	30	-	4	max	9

(a) AEMP benchmarks include: 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).
(b) 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 annual average concentration as presented in the EAR (De Beers 2002).
(c) The 2011 predictions are based on maximum predicted whole-lake average concentrations (De Beers 2011b). The whole-lake average concentration calculations excluded the northwest arm.
(d) Observed concentrations within the 2012 reporting period (January 1, 2012 to September 30, 2012). **Bold** values are above relevant benchmarks. 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 2011 model predictions (De Beers 2011b) are underlined.
(e) Range in average = minimum and maximum whole-lake average concentrations, which excludes northwest arm stations. The exception is phosphorus, for which the range refers to the open-water whole-lake averages.
(f) The ammonia WQG is pH and water temperature dependent. The range of the WQG shown is based on a range of laboratory pH from 6.8 to 7.7 and a range of water temperature from 0.9°C to 18.1°C, which were observed in Snap Lake during the 2012 reporting period. The WQG was calculated based on an individual pH and water temperature for each sample with the final value expressed as ammonia as nitrogen.
(g) The aluminum WQG is pH dependent. The WQG shown is based on a range of pH from 6.8 to 7.7, which was observed in Snap Lake during the 2012 reporting period. The WQG was calculated based on the individual pH for each sample.
(h) The lead and nickel WQGs are hardness dependent. The range of the WQGs shown were based on a range of hardness from 25.6 to 186 mg/L, which was observed in Snap Lake during the 2012 reporting period. The WQG was calculated based on the individual hardness for each sample.

N = nitrogen; - = not applicable; <= less than; ≤ = less than or equal to; 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.

One cadmium result from Snap Lake was above the maximum whole-lake annual average EAR prediction of 0.058 micrograms per litre (µg/L) (De Beers 2002) and above the 2011 Water Licence Renewal Application maximum whole-lake average model prediction of 0.039 µg/L (De Beers 2011b). This sample was collected from the diffuser area. Cadmium concentrations in Snap Lake between 2004 and 2011 have occasionally exceeded maximum whole-lake average model predictions. However, whole-lake average concentrations for cadmium have remained below model predictions. The elevated cadmium results are likely attributable to sample contamination or other isolated issues rather than treated effluent exposure, because there was no temporal trend for cadmium (Figure A6-31 and Section 3.4.4) and concentrations were not correlated with conductivity (Appendix Table 3F-2).

One hexavalent chromium result from Snap Lake was above the maximum whole-lake annual average EAR prediction of 0.8 µg/L (De Beers 2002). This sample was collected from the diffuser area and had a concentration of 1.2 µg/L (Table 3-16). The remainder of the samples collected in 2012 in Snap Lake had hexavalent chromium concentrations below the best available laboratory detection limit (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 value detected at one station in 2012 was near the DL, it is an anomaly and not an indication of hexavalent chromium in groundwater seepage to the underground Mine as outlined in the EAR. A recommendation from the 2011 Water Licence Renewal Application model results report (De Beers 2011b) was to monitor the treated effluent discharge to Snap Lake for hexavalent chromium; this monitoring was initiated in late 2012. All samples collected to date have contained hexavalent chromium at concentrations below the DL of 1 µg/L (Seto 2013).

Three manganese results from Snap Lake were above the maximum whole-lake annual average EAR prediction of 19 µg/L (De Beers 2002). Similar to previous years (De Beers 2010, 2011a), these samples were collected from the northwest arm (i.e., SNAP20B). Based on the spatial pattern of treated effluent exposure (Section 3.4.5), the source of the manganese measured in the northwest arm was not likely the treated effluent from the WTP. There was no temporal trend for total manganese in the northwest arm (Section 3.4.4). The highest concentrations of parameters associated with treated effluent are typically measured closer to the diffuser. Elevated manganese concentrations were more likely related to lower dissolved oxygen (DO) at this location (Section 3.4.5; Figure 3-10), and the reduction of manganese to the more soluble form under such conditions. Under aerobic conditions, manganese is stable in its oxidized form and is highly insoluble. However, when low DO conditions are prevalent, manganese is reduced to the more soluble form, Mn (II), which can result in elevated dissolved concentrations in the water column. Manganese concentrations have not increased over time and were not correlated with conductivity (Section 3.4.4).

Toxicity Data Summary

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 April and three were collected and tested 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 A5).

Seasonal Kendall Test

The results of the Seasonal Kendall test for parameters representative of the major parameter groups (TDS, nutrients [e.g., TP and total nitrogen], and metals [molybdenum and strontium]), are summarized in Table 3-17. Fluoride was tested because maximum concentrations in 2012 were above the CCME WQG of 0.12 mg/L (Table 3-13) and above the maximum whole-lake average 2011 model prediction of 0.14 mg/L (Table 3-16). Maximum concentrations of chloride and nitrate in 2012 were also above CCME WQGs, but trends for chloride and nitrate were implicit in trends for TDS and total nitrogen. Cadmium was tested because maximum concentrations in 2012 were above the maximum whole-lake annual average EAR prediction of 0.058 µg/L and above the maximum whole-lake average 2011 model prediction of 0.039 µg/L (Table 3-16). Manganese was tested because maximum concentrations in 2012 were above the maximum whole-lake annual average EAR prediction of 19 µg/L.

The test identified similar temporal trends as those identified through the screening and visual evaluation of temporal plots, with the exception of TP and total manganese. Increasing trends were evident at all stations at all of the tested depths for calculated TDS, total nitrogen, fluoride, total molybdenum, and total strontium. The Seasonal Kendall test identified a decreasing trend for TP concentrations at the bottom depth of the diffuser station and at the mid-field station, and an increasing trend for total manganese concentrations at the surface and mid depths at the diffuser station. Results from the Seasonal Kendall test for cadmium indicated that cadmium concentrations are neither increasing nor decreasing in Snap Lake.

Table 3-17 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 Dissolved Solids, Calculated (as an upward trend)	Diffuser (SNP02-20e)	Bottom	47	6.810	0.000	↑
		Mid	47	8.765	0.000	↑
		Surface	45	8.171	0.000	↑
	Near-field (SNAP05)	Mid	33	7.439	0.000	↑
	Mid-field (SNAP09)	Mid	34	7.797	0.000	↑
	Far-field (SNAP08)	Mid	36	7.850	0.000	↑
	Northwest Arm (SNAP02A)	Mid	25	6.017	0.000	↑
Total Phosphorus (as a two-sided trend)	Diffuser (SNP02-20e)	Bottom	47	-2.680	0.007	↓
		Mid	47	-1.950	0.051	-
		Surface	45	-1.588	0.112	-
	Near-field (SNAP05)	Mid	33	-1.848	0.065	-
	Mid-field (SNAP09)	Mid	34	-4.044	0.000	↓
	Far-field (SNAP08)	Mid	36	-1.634	0.102	-
	Northwest Arm (SNAP02A)	Mid	25	-0.417	0.676	-
Total Nitrogen (as an upward trend)	Diffuser (SNP02-20e)	Bottom	47	7.994	0.000	↑
		Mid	47	8.112	0.000	↑
		Surface	45	8.037	0.000	↑
	Near-field (SNAP05)	Mid	31	6.568	0.000	↑
	Mid-field (SNAP09)	Mid	31	6.627	0.000	↑
	Far-field (SNAP08)	Mid	36	7.280	0.000	↑
	Northwest Arm (SNAP02A)	Mid	25	0.365	0.358	-
Fluoride (as a two-sided trend)	Diffuser (SNP02-20e)	Bottom	47	4.408	0.000	↑
		Mid	46	5.425	0.000	↑
		Surface	45	5.861	0.000	↑
	Near-field (SNAP05)	Mid	33	4.574	0.000	↑
	Mid-field (SNAP09)	Mid	34	5.619	0.000	↑
	Far-field (SNAP08)	Mid	36	6.239	0.000	↑
	Northwest Arm (SNAP02A)	Mid	27	2.001	0.045	↑
Total Cadmium (AITF) (as a two-sided trend)	Diffuser (SNP02-20e)	Bottom	47	0.936	0.349	-
		Mid	47	0.000	1.000	-
		Surface	45	-0.115	0.908	-
	Near-field (SNAP05)	Mid	31	-1.274	0.203	-
	Mid-field (SNAP09)	Mid	31	-1.067	0.286	-
	Far-field (SNAP08)	Mid	32	-1.433	0.152	-
	Northwest Arm (SNAP02A)	Mid	27	1.947	0.051	-

Table 3-17 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 Manganese (as a two-sided trend)	Diffuser (SNP02-20e)	Bottom	47	-0.855	0.393	-
		Mid	47	2.339	0.019	↑
		Surface	45	2.826	0.005	↑
	Near-field (SNAP05)	Mid	31	1.468	0.142	-
	Mid-field (SNAP09)	Mid	31	1.299	0.194	-
	Far-field (SNAP08)	Mid	36	-0.482	0.629	-
	Northwest Arm (SNAP02A)	Mid	25	0.137	0.891	-
Total Molybdenum (as an upward trend)	Diffuser (SNP02-20e)	Bottom	47	7.734	0.000	↑
		Mid	47	7.811	0.000	↑
		Surface	45	7.833	0.000	↑
	Near-field (SNAP05)	Mid	31	6.835	0.000	↑
	Mid-field (SNAP09)	Mid	31	7.035	0.000	↑
	Far-field (SNAP08)	Mid	36	7.235	0.000	↑
	Northwest Arm (SNAP02A)	Mid	25	4.103	0.000	↑
Total Strontium (as an upward trend)	Diffuser (SNP02-20e)	Bottom	47	5.556	0.000	↑
		Mid	47	8.364	0.000	↑
		Surface	45	8.171	0.000	↑
	Near-field (SNAP05)	Mid	33	6.443	0.000	↑
	Mid-field (SNAP09)	Mid	31	7.397	0.000	↑
	Far-field (SNAP08)	Mid	36	7.874	0.000	↑
	Northwest Arm (SNAP02A)	Mid	25	6.381	0.000	↑

Note: The Seasonal Kendall Test was run using SYSTAT 13.1.00.5 (SYSTAT 2009); ↓ = a decreasing trend; ↑ = an increasing trend; - = no significant increasing or decreasing trend n = sample count.

(a) The critical *Z*-values associated with a one-sided 95% confidence interval are -1.64 and 1.64. 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 less than -1.64 for a downward trend test or less than 1.64 for an upward trend test (one-sided tests), the *P*-value will be greater than 0.05 and the test concludes that no significant increasing or decreasing trend exists in the data. 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.

AITF = Alberta Innovates Technology Futures; % = percent; n = number of samples.

Dissolved Oxygen

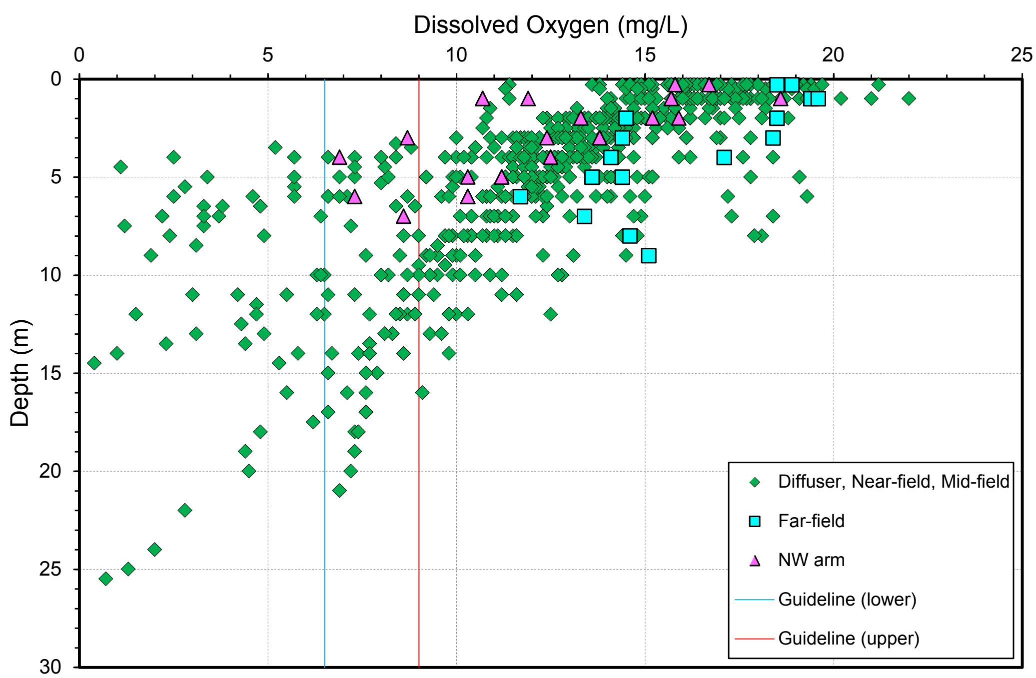
Vertical profiles of DO concentration during ice-cover between 1999 and 2012 are shown in Figure 3-22 (panels a to i). Profile data collected before treated effluent discharge to Snap Lake (1999 to 2004) were combined onto one graph (panel a), and data collected during the period of treated effluent discharge are presented by year (panels b to i). A greater number of deeper stations were sampled in 2006 to 2012 compared to 2005 (Figure 3-22, panels b to h) 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 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, panel a). Before the discharge began (1999 to 2004), DO concentrations decreased with depth to near 0 mg/L at deeper near-field, far-field, and northwest arm stations during ice-covered conditions. In general, between 2005 and 2012, near-bottom DO concentrations at near-field 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 2012 at this location. Low oxygen and anoxic conditions were observed near the bottom at some locations in the northwest arm in 2007, 2009, 2010, 2011, and 2012 (Figure 3-22, panels d to i).

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

a. 1999 to 2004 – Pre-discharge



b. 2005

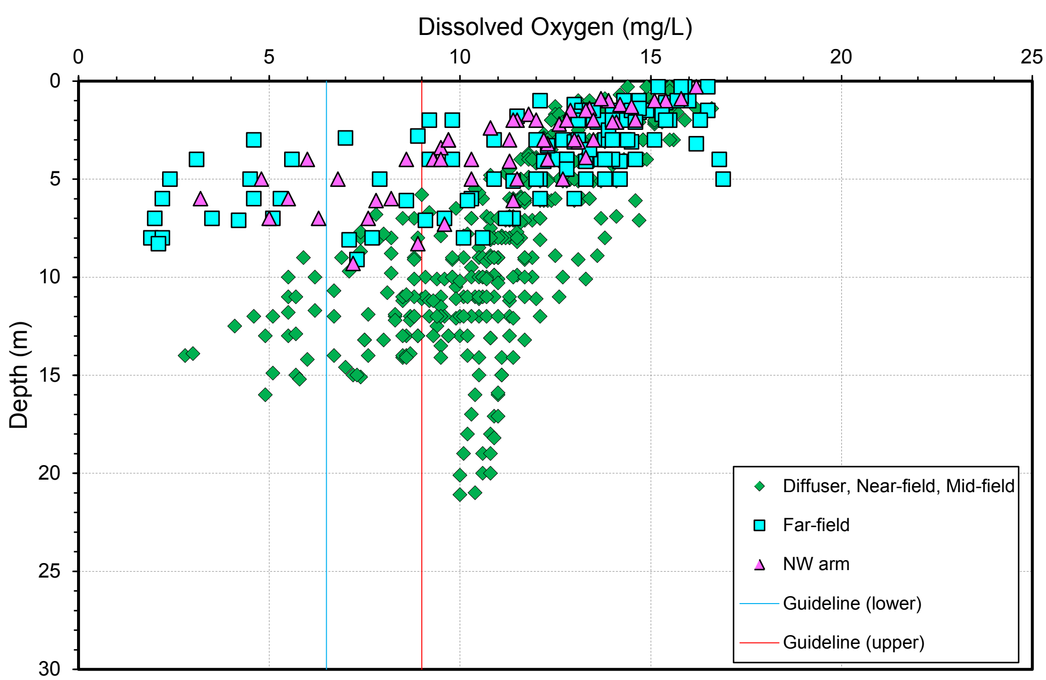
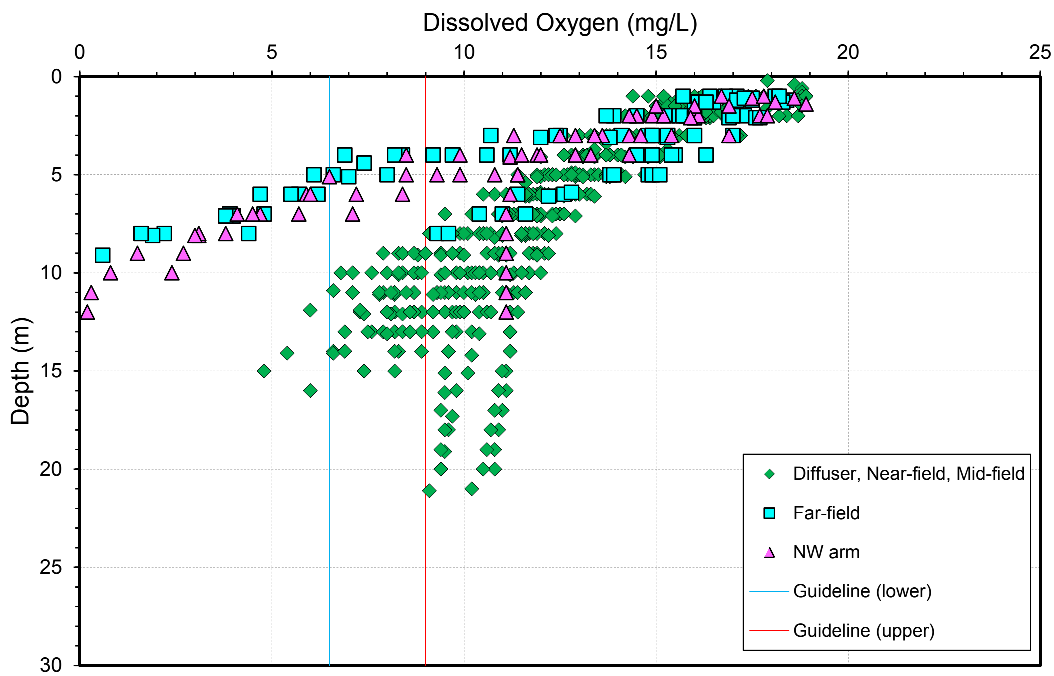


Figure 3-22 Dissolved Oxygen Concentrations During Ice-Covered Conditions, 1999 to 2012

c. 2006



d. 2007

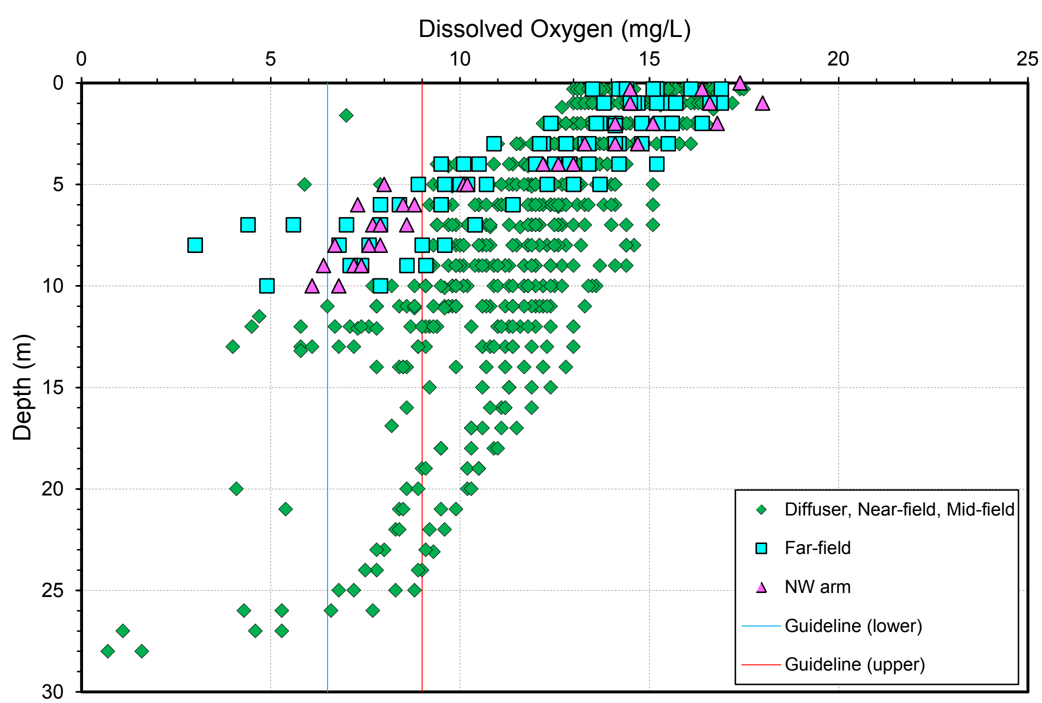
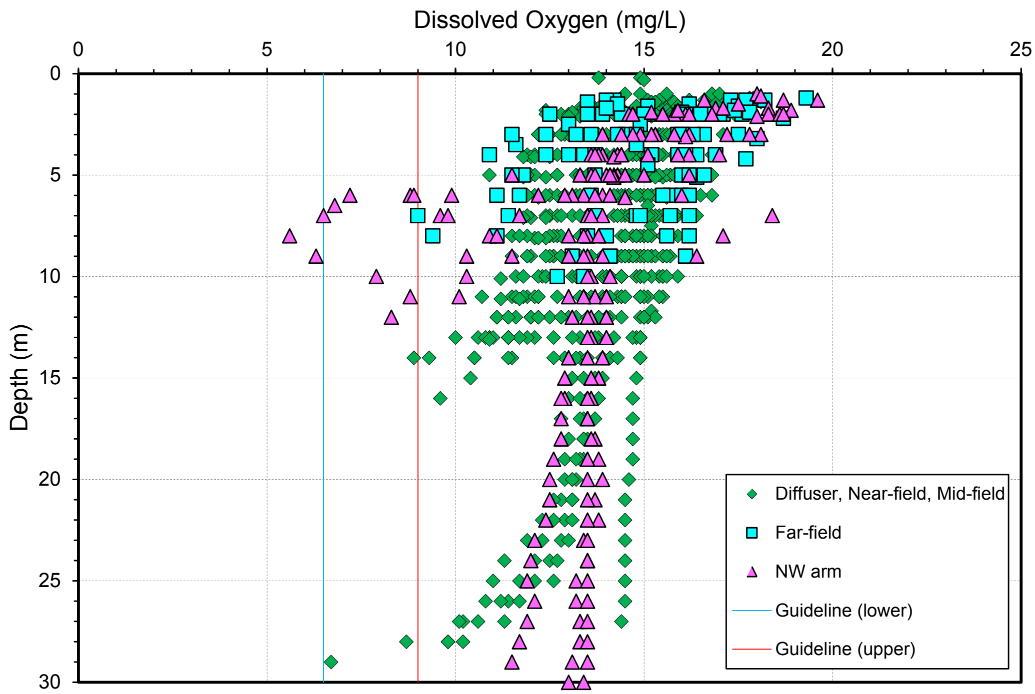


Figure 3-22 Dissolved Oxygen Concentrations During Ice-Covered Conditions, 1999 to 2012

e. 2008



f. 2009

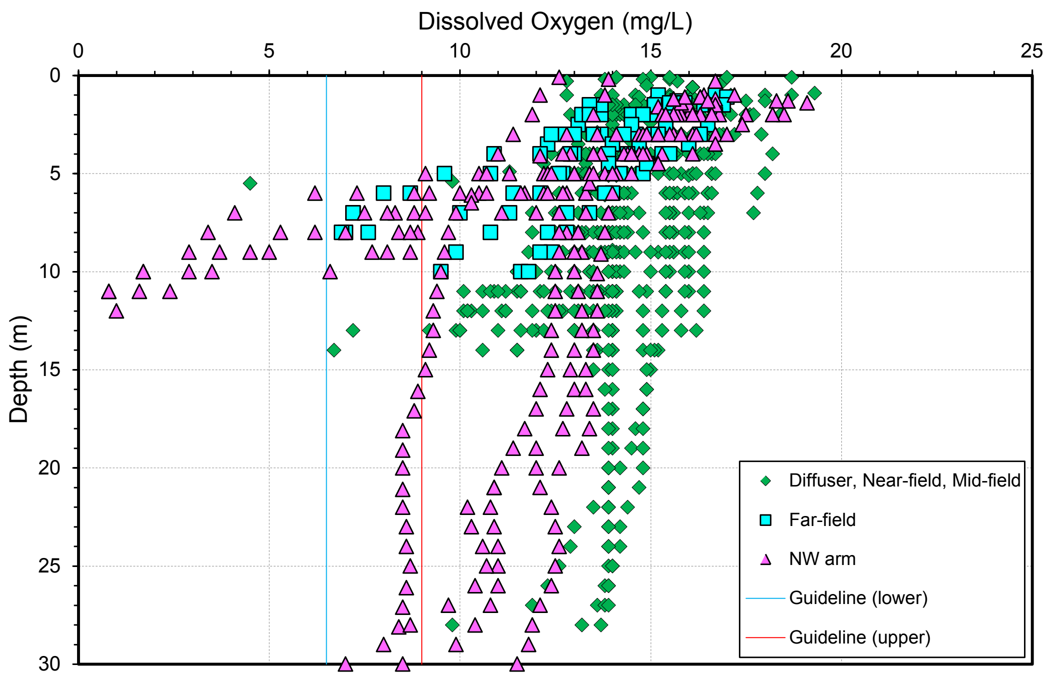
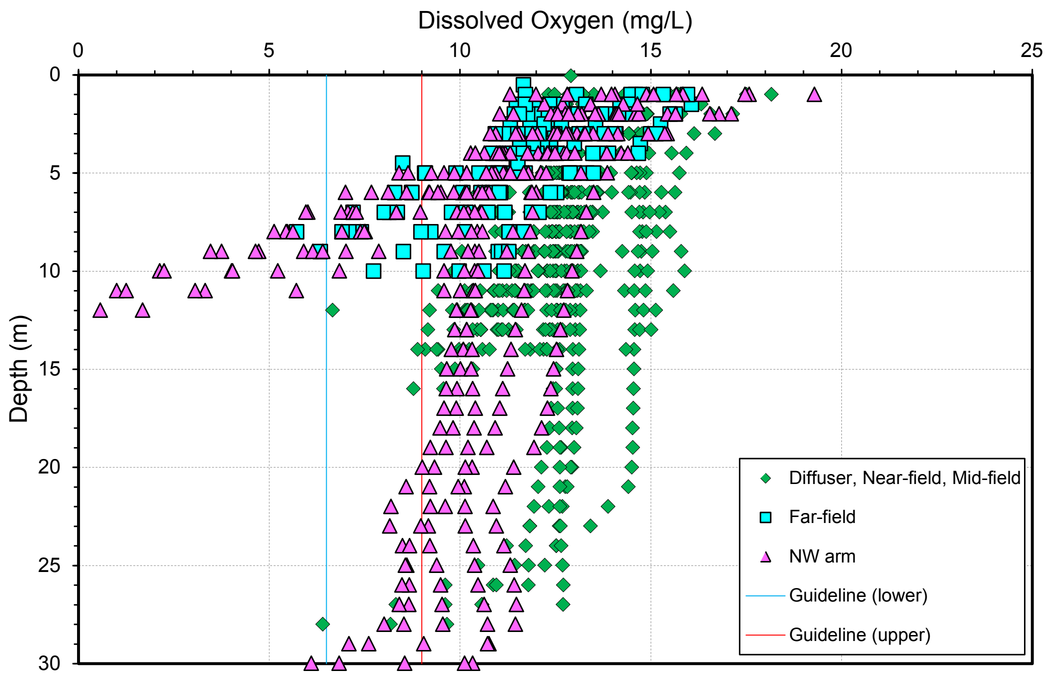


Figure 3-22 Dissolved Oxygen Concentrations During Ice-Covered Conditions, 1999 to 2012

g. 2010



h. 2011

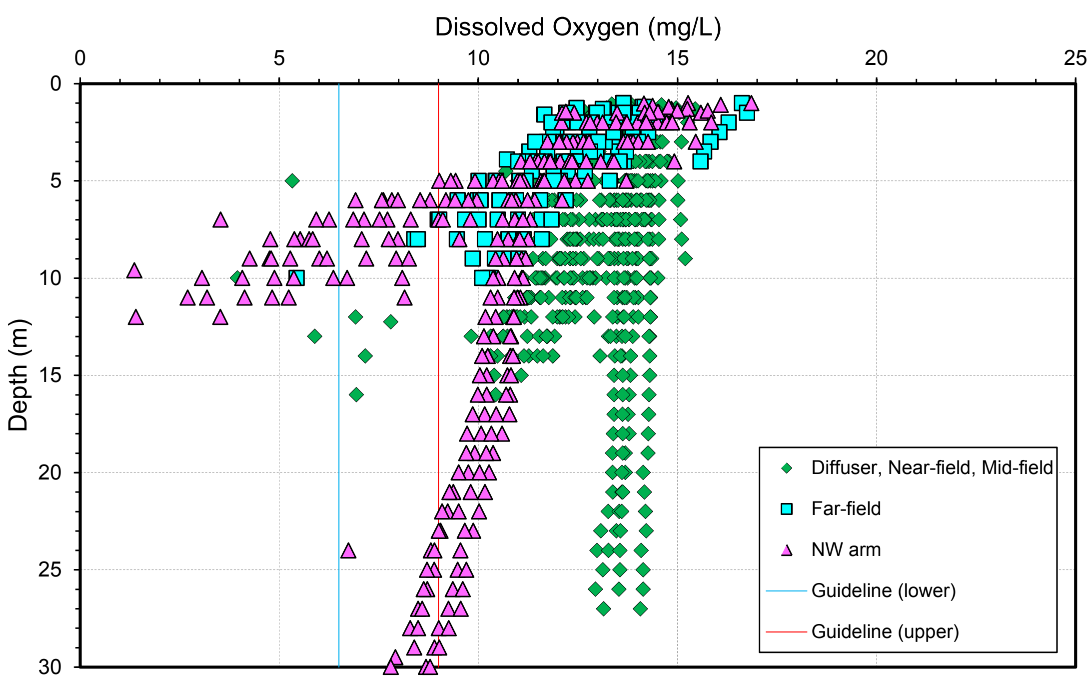
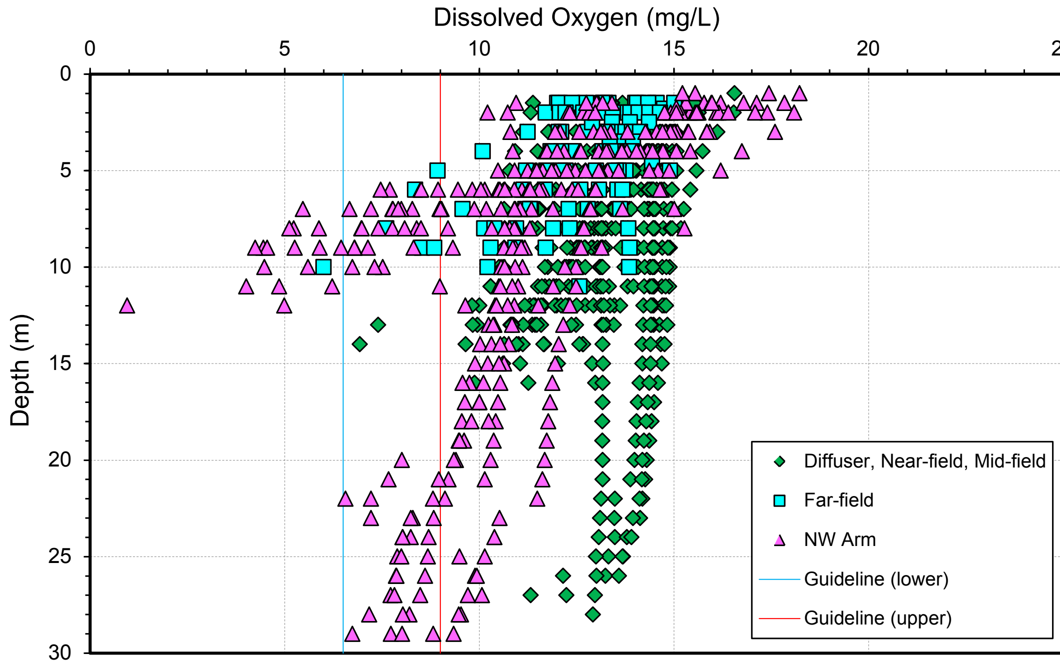


Figure 3-22 Dissolved Oxygen Concentrations During Ice-Covered Conditions, 1999 to 2012

i. 2012



NW Arm = Northwest Arm; data shown are from representative diffuser, near-field, mid-field, far-field, and NW arm stations within 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 = metres.

3.4.4.2 Summary of Key Question 3

In 2012, the following parameters increased to concentrations above the Snap Lake normal range (i.e., baseline mean ± two standard deviations) and reference lake (Northeast Lake and Lake 13) concentrations in at least one area of Snap Lake:

- TDS, total alkalinity, 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).

Whole-lake average TDS concentrations in Snap Lake in 2012 were higher than the 2011 Water Licence Renewal Application model predictions. Measured whole-lake average concentrations of nitrate in Snap Lake have been increasing since 2005, consistent with EAR and recent modelling predictions. In 2012, concentrations of nitrate were below maximum EAR predictions.

In 2012, increases in surface and bottom water DO concentrations were measured over the winter 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.

Toxicity test results from three diffuser samples collected in April and three collected in September showed no adverse effects for any test endpoints. Algal growth was stimulated in all samples; however, with the degree of stimulation increasing at higher sample concentrations (Appendix A5).

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 horizontal, vertical, and seasonal patterns in Snap Lake water quality for field parameters, total dissolved solids, major ions, nutrients, and metals. Where patterns existed, the potential for Mine-related causes were assessed.

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 the reference lakes 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. Spatial and seasonal plots for all laboratory parameters are presented in Appendix 3G.

Finally, water quality data for the AEMP downstream station, KING01, were reviewed to identify potential changes in water quality at a station located 25 km downstream of Snap Lake. A Downstream Lakes Special Study was conducted in three lakes (DSL1, DSL2, and Lac Capot Blanc) immediately downstream of Snap Lake to delineate the spatial extent of the treated effluent plume and assess conditions in 2012. Detailed information on the Downstream Lakes Special Study is presented in Section 12.2; however, a short summary is provided in this section.

3.4.5.1 Spatial Patterns and Seasonal Variation for Snap Lake

Field Parameters

Conductivity

In 2012, conductivity in Snap Lake ranged from 90 µS/cm in the northwest arm in August to 558 µS/cm at diffuser station SNP02-20e in April (Appendix 3B; Table 3B-1), higher than the range observed in 2011 (De Beers 2012b). Conductivity is a reliable field indicator of TDS, which is elevated in the treated effluent relative to Snap Lake waters. The close relationship between conductivity and TDS in Snap Lake from 2004 to 2012 is illustrated in Figure 3-23.

Spatial variability within the main basin of Snap Lake in 2012, excluding the northwest arm, was consistent with recent years (De Beers 2010, 2011a, 2012b). Field conductivity measurements in the different lake areas, between the main basin and northwest arm, were consistent with exposure to the treated effluent based on proximity to the discharge and hydraulic connectivity in 2012 (Figures 3-24 to 3-28). That is, the order of measured conductivity from highest to lowest in September was: diffuser stations, near-field area, mid-field area, far-field area, and northwest arm (Figure 3-28). The highest conductivity in 2012 was measured near the diffuser towards the end of the ice-covered season (e.g., April; Figures 3-29 to 3-31); however, the differences in conductivity between areas in the main basin of the lake are becoming less distinguishable (Figures 3-24 to 3-28).

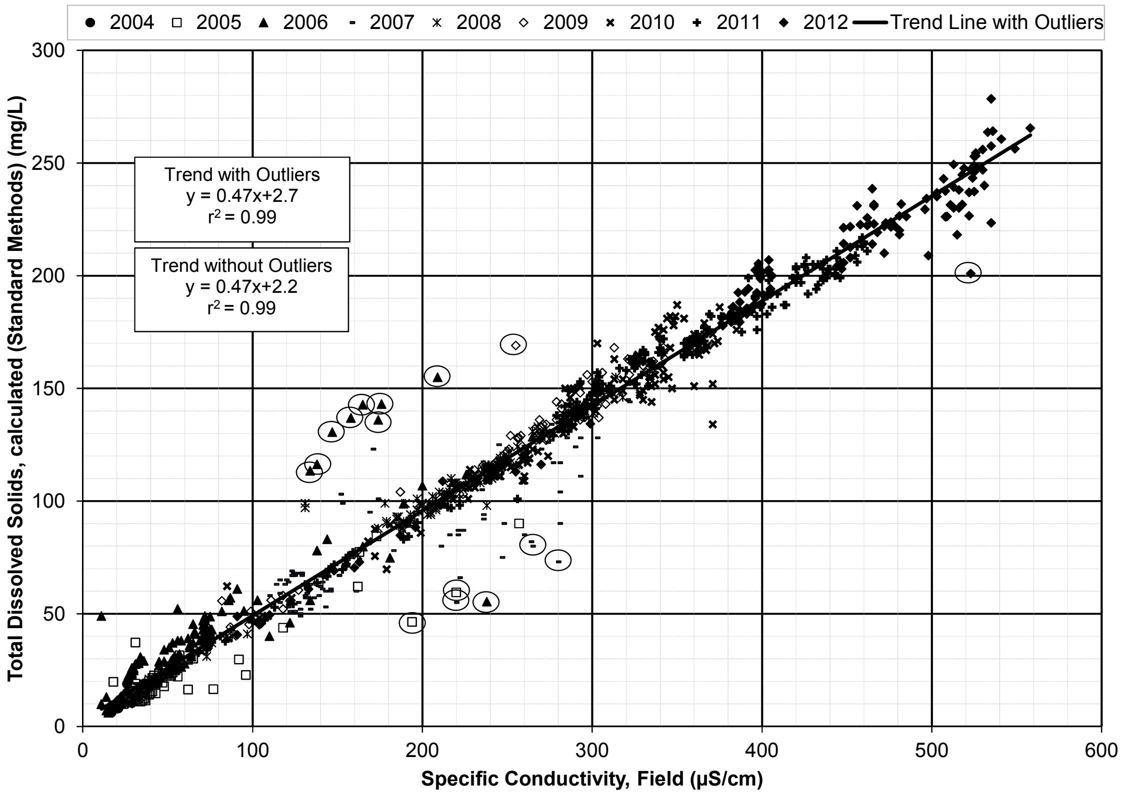
At the deepest diffuser station (i.e., SNP 02-20e), conductivity increased slightly with depth during the February and April programs (Figures 3-25 and 3-26, respectively). Conductivity in April at the shallower diffuser stations (i.e., SNP 02-20d and SNP 2-20f) increased from the surface to a depth of about 5 to 10 m, and then remained relatively consistent to the bottom of the lake (Figure 3-26). Conductivity at the near-field stations also increased from the surface to a depth of about 5 to 10 m, but then decreased again with depth, indicating that plume concentrations were highest at mid-depth in the near-field in April (Figure 3-26). Higher conductivity at mid-depth may be due to the influence from the diffuser, which has ports that discharge treated effluent away from the bottom of the lake. Historically, the higher density of the treated effluent plume relative to the lake water has 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 2012 the plume appeared to be situated mid-column rather than sinking to the bottom as observed prior to 2009.

In 2012, conductivity in the northwest arm was lower compared to other areas of Snap Lake, consistent with historical spatial trends since the Mine began operating (De Beers 2005a, 2006, 2007a, 2008a, 2009, 2010, 2011a, 2012b). 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 3-24 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 also show that the treated effluent is situated near the bottom of the water column in the northwest arm, particularly at stations SNAP 23 and SNAP 02A (Figures 3-24 and 3-28).

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) but becomes fully mixed by the end of the open-water season in September (Figure 3-28). At the deepest diffuser station (SNP 02-20e), conductivity similar to late winter was measured in July, indicating the water column was not yet completely mixed (Figure 3-27). During the September 2012 field program, complete vertical mixing occurred at all stations in the main basin, and the plume was well-mixed vertically (Figures 3-28 and 3-32). The lack of a discernible vertical gradient at 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 2012, conductivity measured in Northeast Lake during ice-covered and open-water conditions was substantially lower compared to all measured conductivity in Snap Lake (Figures 3-24 to 3-28). A slight vertical gradient in conductivity in the surface water zone in Northeast Lake was evident 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 not noticeable in most of Snap Lake due to the elevated conductivity associated with treated effluent.

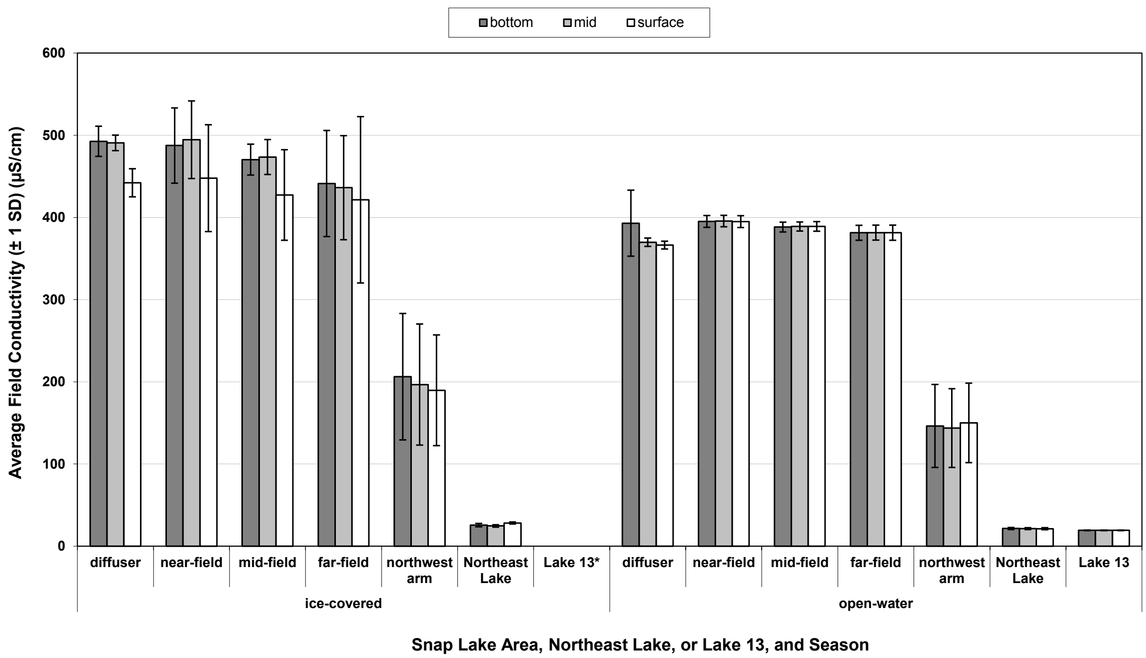
Figure 3-23 Relationship Between Field Conductivity and Total Dissolved Solids in Snap Lake, 2004 to 2012



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

Note: Circled data points are outliers

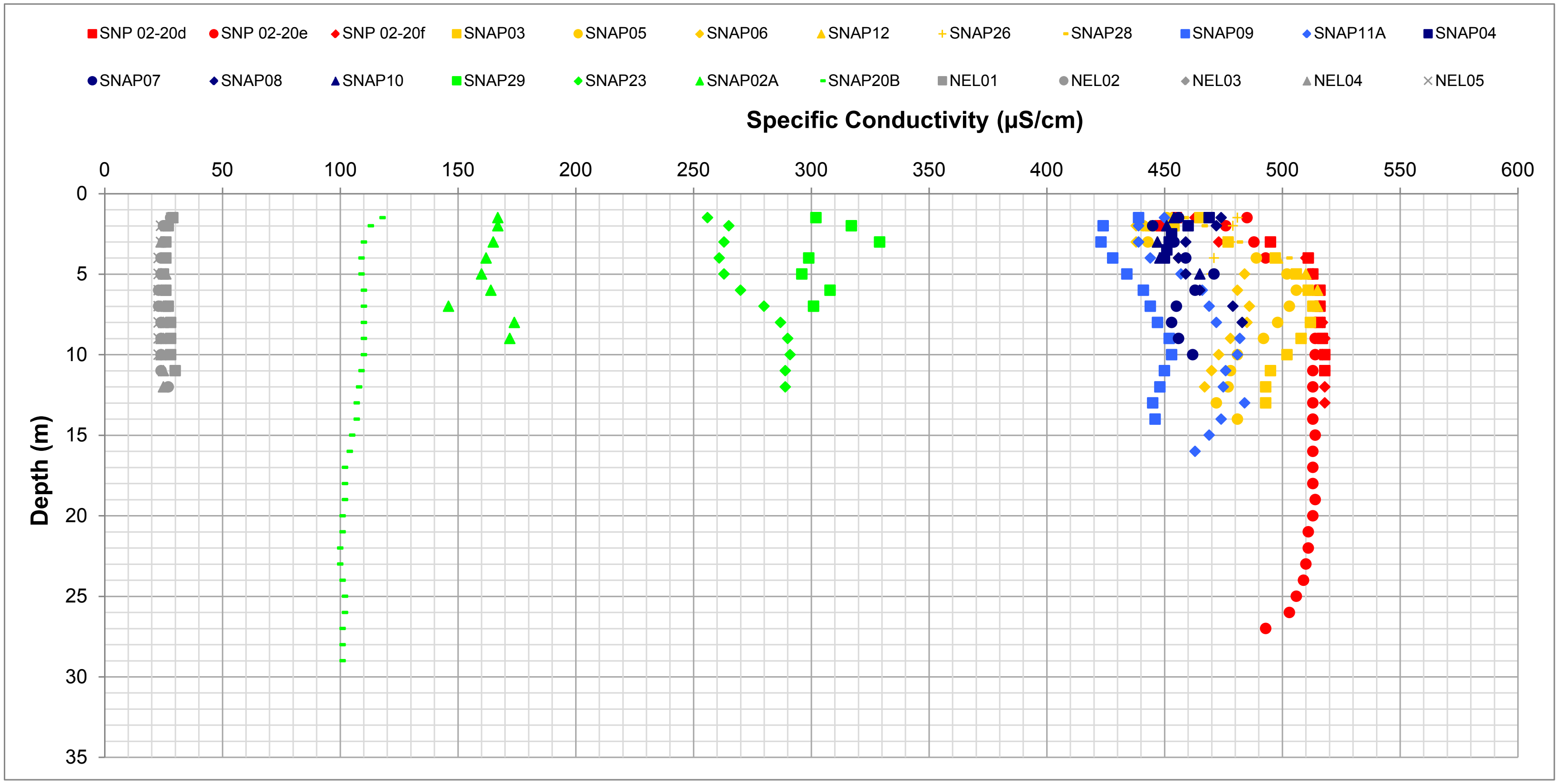
Figure 3-24 Average of Field Conductivity (± 1 SD) in Snap Lake, Northeast Lake, and Lake 13, 2012



* Water samples were not collected from Lake 13 during the ice-covered season.

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

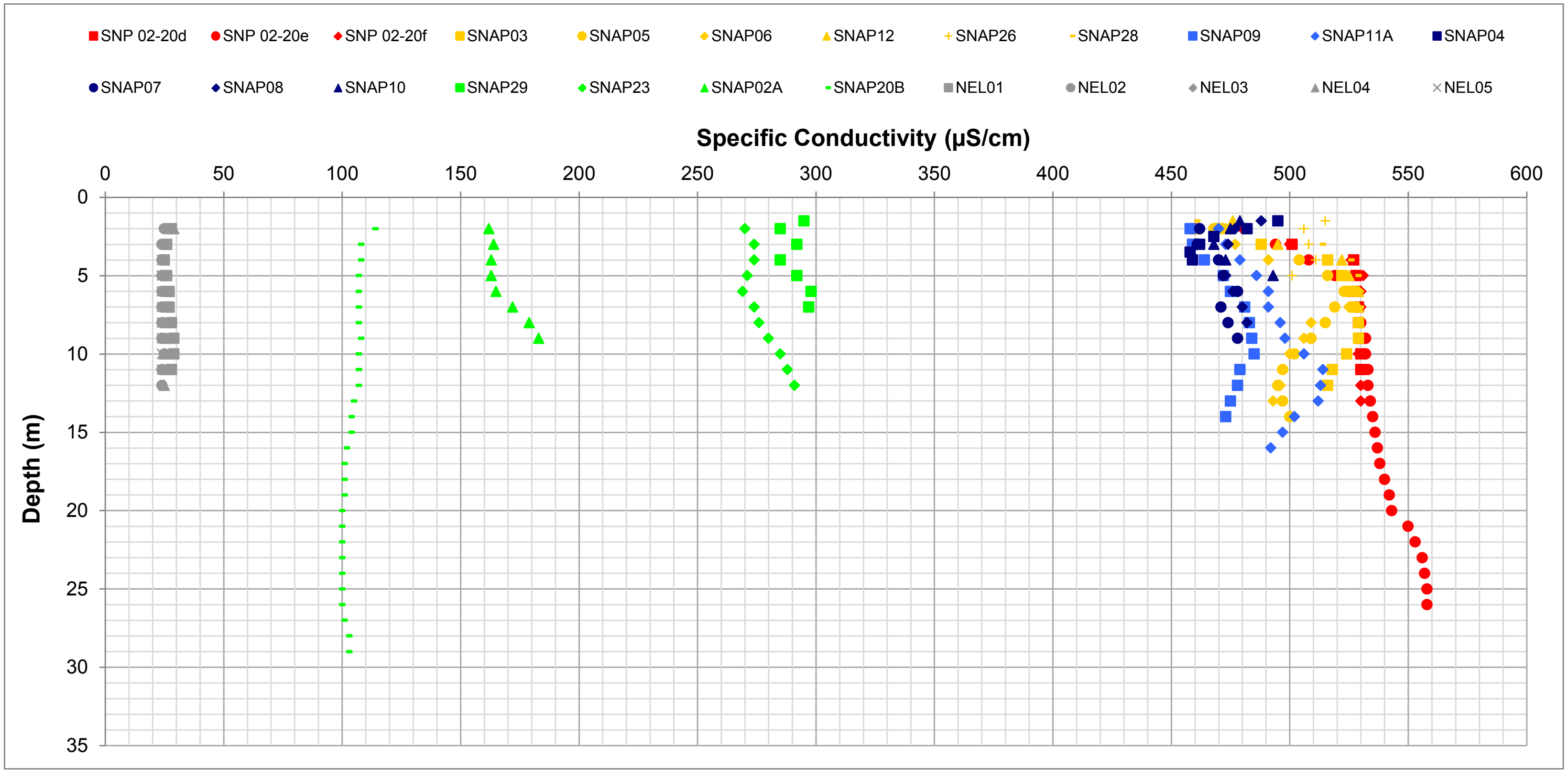
Figure 3-25 Field Conductivity Profiles in Snap Lake, February 2012



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.

m = metres; µS/cm = microSiemens per centimetre.

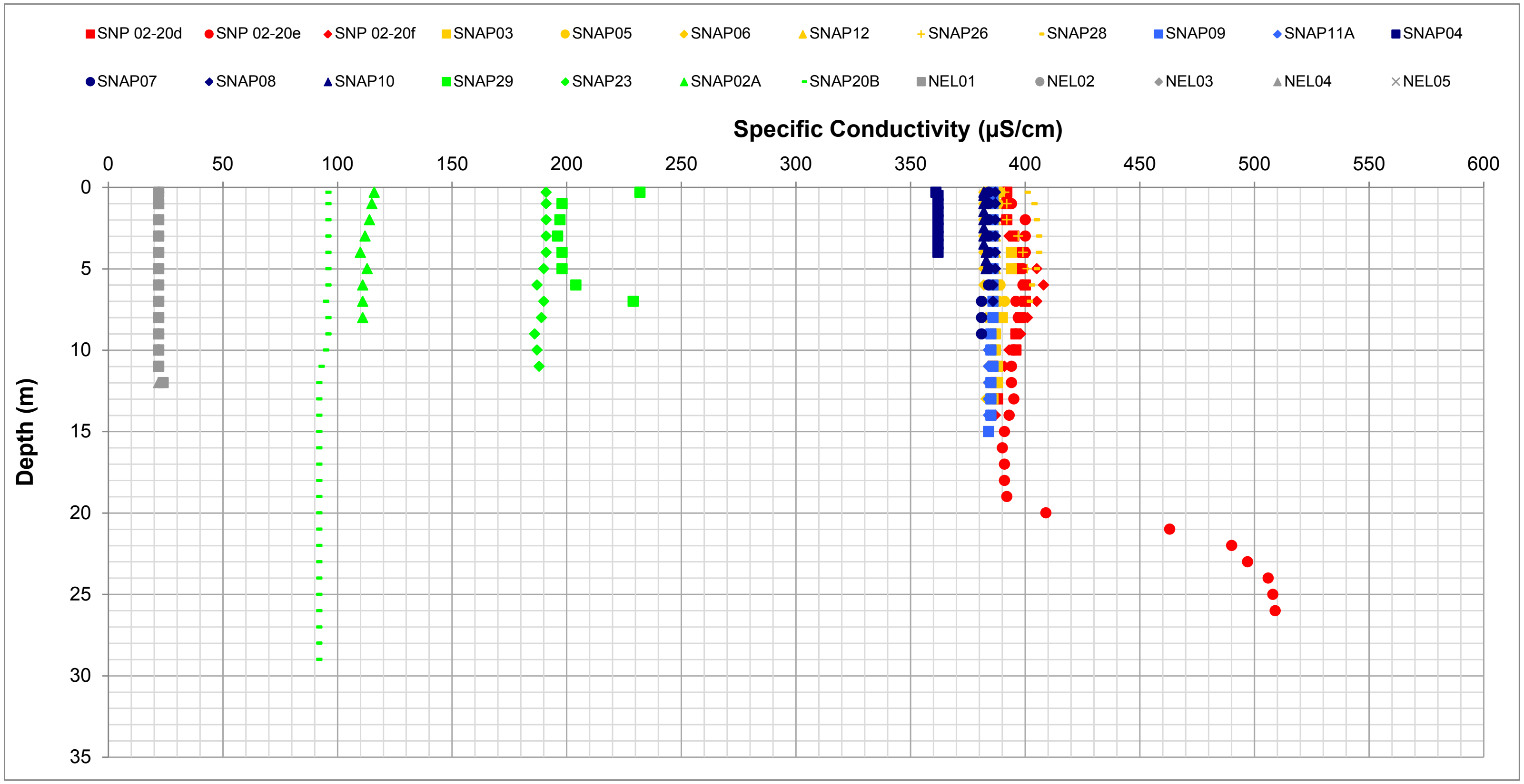
Figure 3-26 Field Conductivity Profiles in Snap Lake and Northeast Lake, April 2012



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.

m = metre; µS/cm = microSiemens per centimetre.

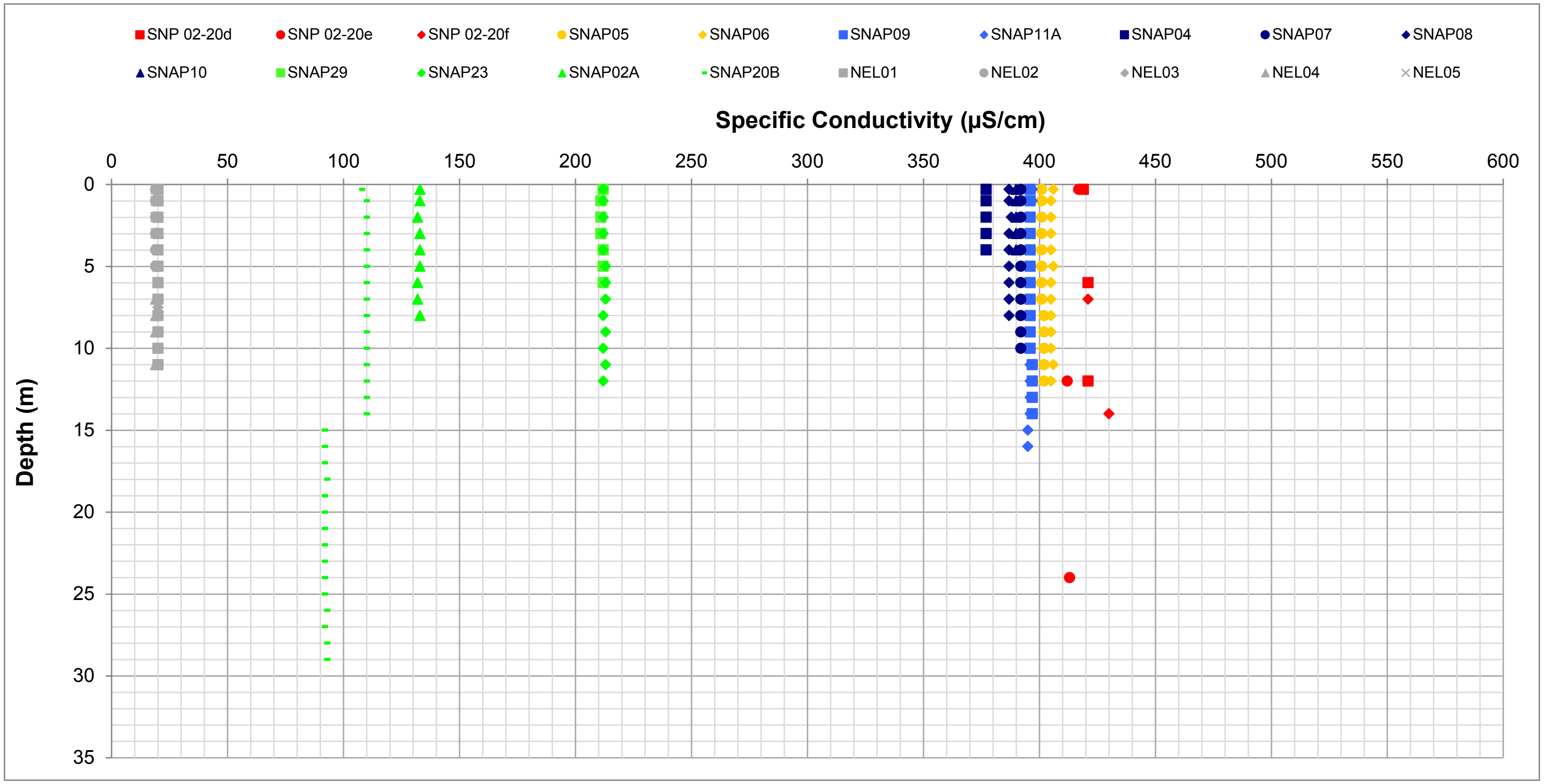
Figure 3-27 Field Conductivity Profiles in Snap Lake and Northeast Lake, July 2012



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.

m = metres; µS/cm = microSiemens per centimetre.

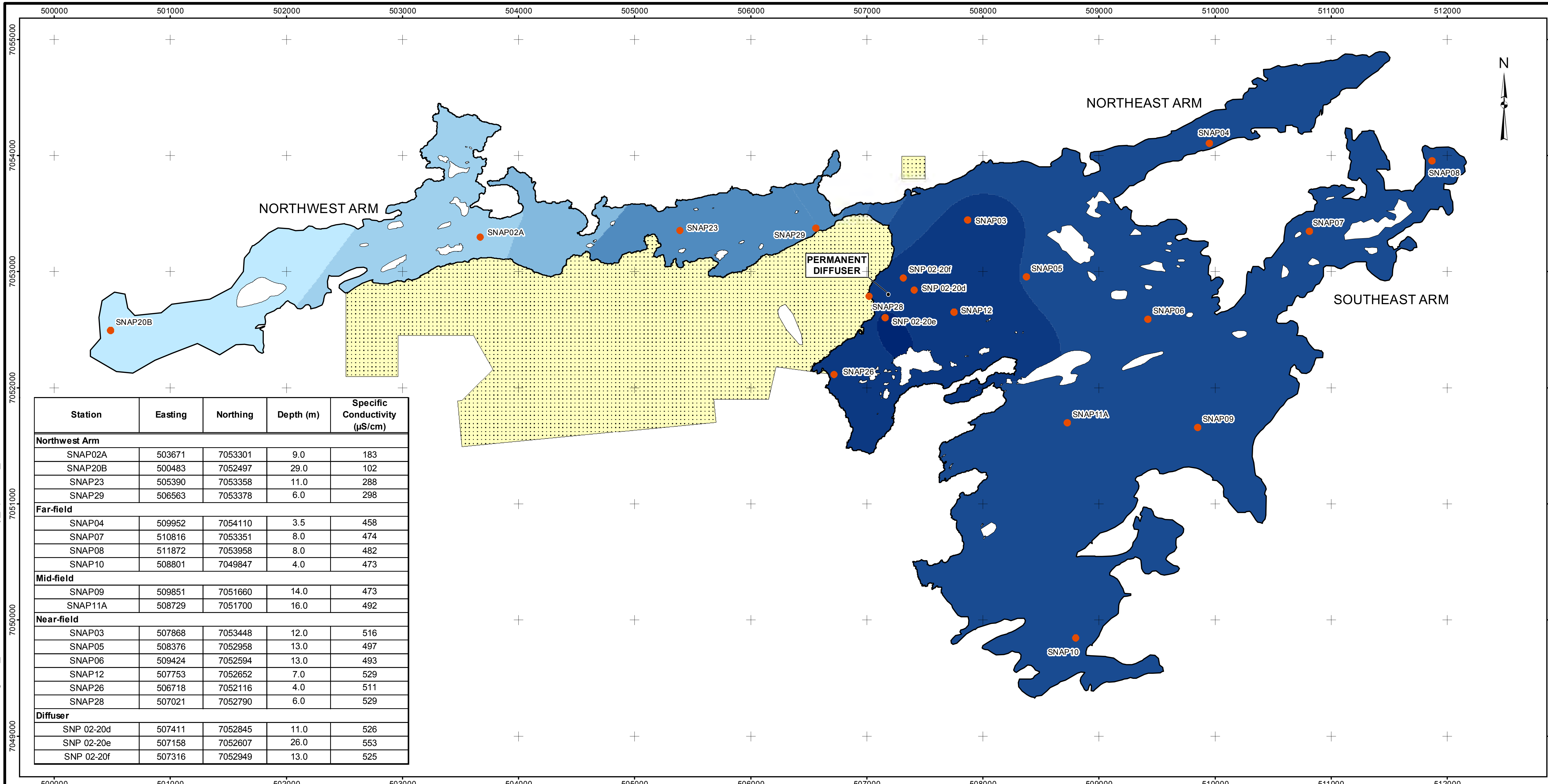
Figure 3-28 Field and Laboratory Conductivity Profiles in Snap Lake and Northeast Lake, September 2012



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 coloured symbols represent the diffuser stations.

All results are for field conductivity, with the exception of the diffuser stations, where only laboratory conductivity data were available at the surface, mid-depth, and bottom. Details are included in Appendix 3A.

m = metre; μS/cm = microSiemens per centimetre.



Station	Easting	Northing	Depth (m)	Specific Conductivity (µS/cm)
Northwest Arm				
SNAP02A	503671	7053301	9.0	183
SNAP20B	500483	7052497	29.0	102
SNAP23	505390	7053358	11.0	288
SNAP29	506563	7053378	6.0	298
Far-field				
SNAP04	509952	7054110	3.5	458
SNAP07	510816	7053351	8.0	474
SNAP08	511872	7053958	8.0	482
SNAP10	508801	7049847	4.0	473
Mid-field				
SNAP09	509851	7051660	14.0	473
SNAP11A	508729	7051700	16.0	492
Near-field				
SNAP03	507868	7053448	12.0	516
SNAP05	508376	7052958	13.0	497
SNAP06	509424	7052594	13.0	493
SNAP12	507753	7052652	7.0	529
SNAP26	506718	7052116	4.0	511
SNAP28	507021	7052790	6.0	529
Diffuser				
SNP 02-20d	507411	7052845	11.0	526
SNP 02-20e	507158	7052607	26.0	553
SNP 02-20f	507316	7052949	13.0	525

1,000 0 1,000
SCALE 1:33,000 METRES

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 µS/cm

NOTE

µS/cm = microSiemens per centimetre
m = metres

REFERENCE

Datum: NAD83 Projection: UTM Zone 12

PROJECT

DE BEERS
GROUP OF COMPANIES

TITLE

BOTTOM CONDUCTIVITY IN SNAP LAKE, APRIL 2012

Golder Associates

PROJECT		13-1349-0001	FILE No.	
DESIGN	KLN	03 Feb. 2012	SCALE AS SHOWN	REV. 0
GIS	GU	09 Apr. 2013	**FIGURE: 3-29**	
CHECK	LMB	19 Apr. 2013		
REVIEW	PMC	18 Apr. 2013		

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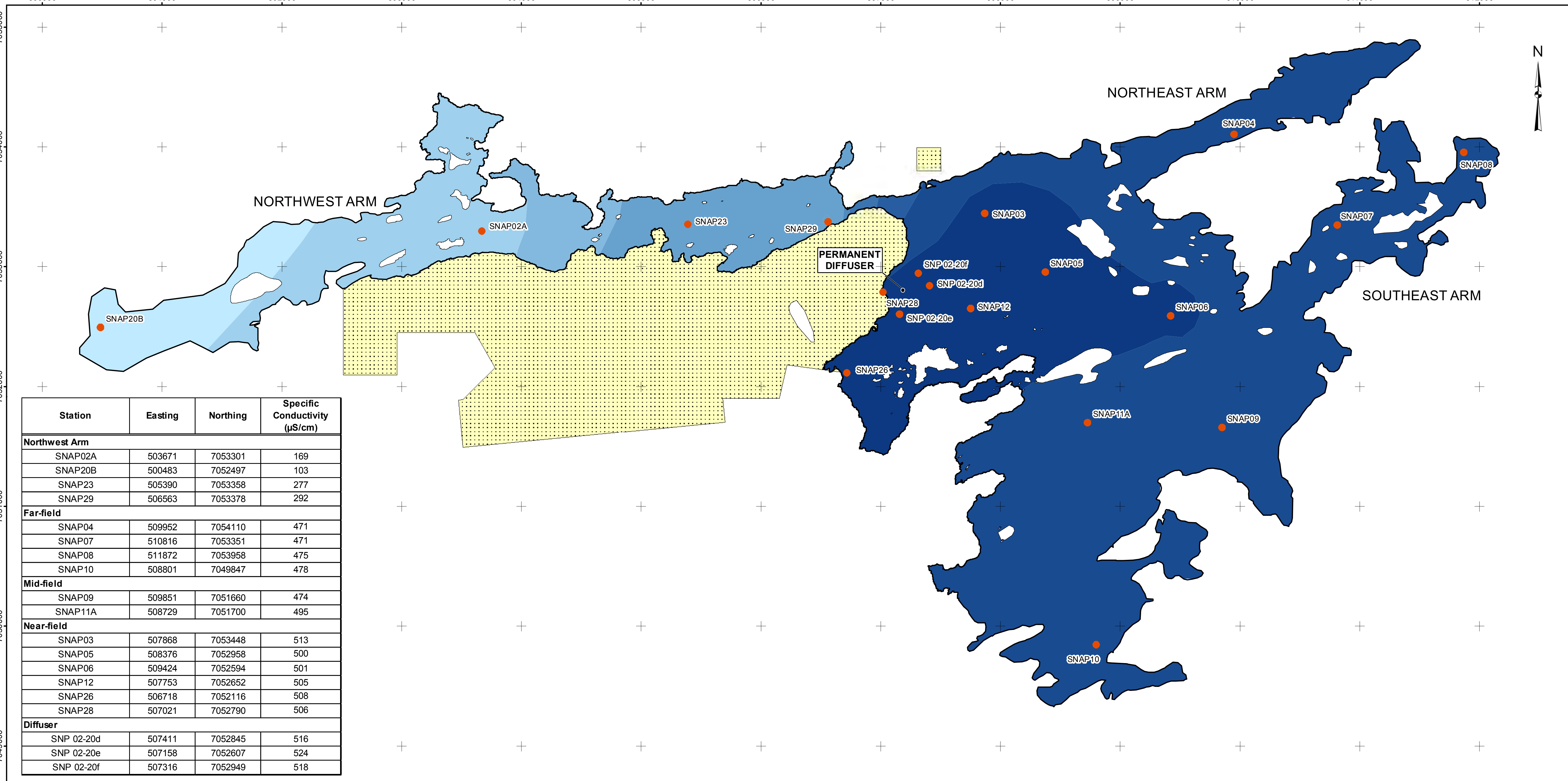

Station	Easting	Northing	Specific Conductivity (µS/cm)
Northwest Arm			
SNAP02A	503671	7053301	169
SNAP20B	500483	7052497	103
SNAP23	505390	7053358	277
SNAP29	506563	7053378	292
Far-field			
SNAP04	509952	7054110	471
SNAP07	510816	7053351	471
SNAP08	511872	7053958	475
SNAP10	508801	7049847	478
Mid-field			
SNAP09	509851	7051660	474
SNAP11A	508729	7051700	495
Near-field			
SNAP03	507868	7053448	513
SNAP05	508376	7052958	500
SNAP06	509424	7052594	501
SNAP12	507753	7052652	505
SNAP26	506718	7052116	508
SNAP28	507021	7052790	506
Diffuser			
SNP 02-20d	507411	7052845	516
SNP 02-20e	507158	7052607	524
SNP 02-20f	507316	7052949	518

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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 µS/cm

1,000 0 1,000

SCALE 1:33,000 METRES

NOTE

µS/cm = microSiemens per centimetre
m = metres

REFERENCE

Datum: NAD83 Projection: UTM Zone 12

PROJECT

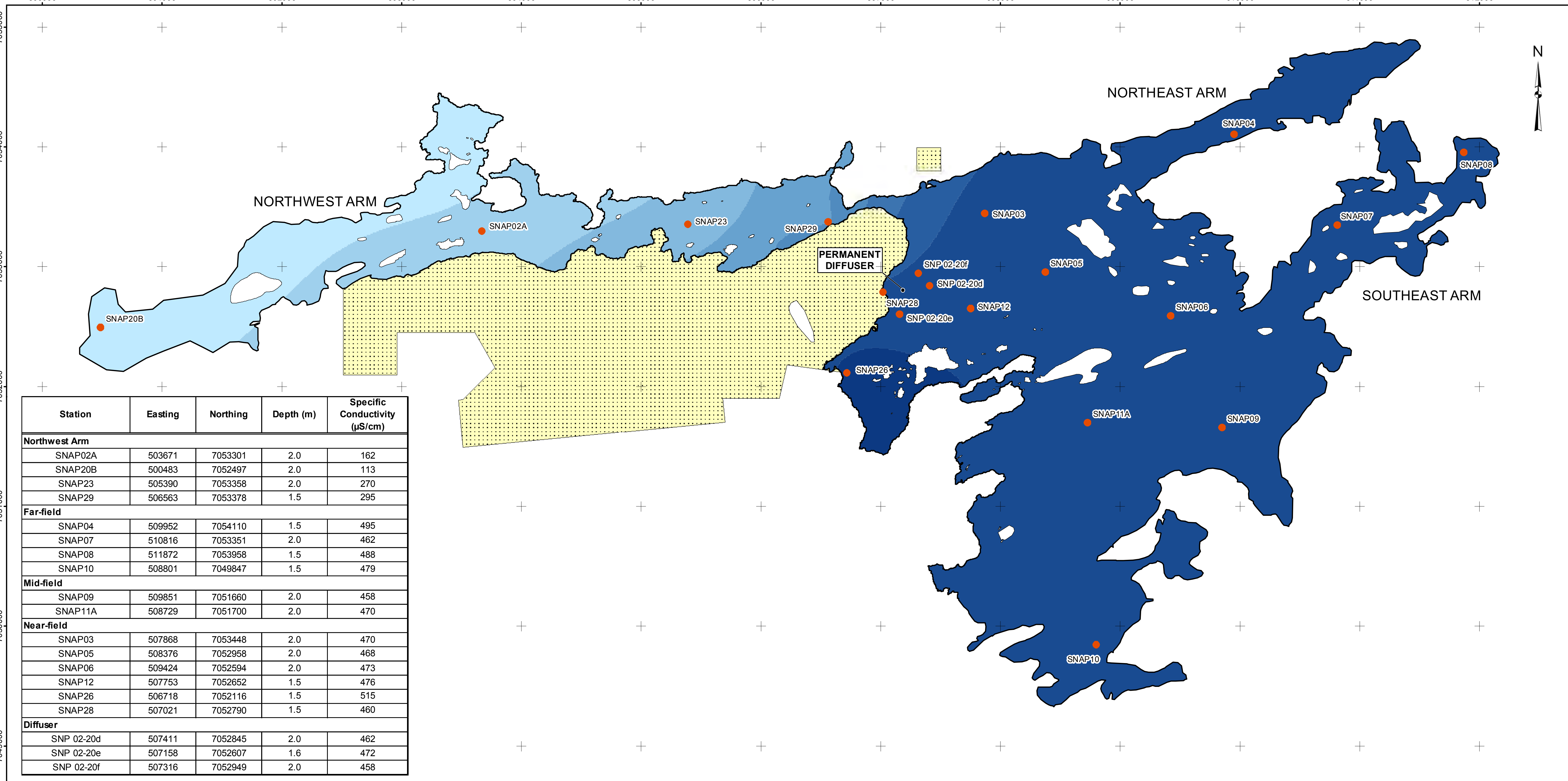
DE BEERS GROUP OF COMPANIES

TITLE

WATER COLUMN MEAN CONDUCTIVITY IN SNAP LAKE, APRIL 2012

Golder Associates

PROJECT		13-1349-0001	FILE No.
DESIGN	CW	03 Feb. 2012	SCALE AS SHOWN REV. 0
GIS	GU	09 Apr. 2013	
CHECK	LMB	19 Apr. 2013	**FIGURE: 3-30**
REVIEW	PMC	18 Apr. 2013	



Station	Easting	Northing	Depth (m)	Specific Conductivity (µS/cm)
Northwest Arm				
SNAP02A	503671	7053301	2.0	162
SNAP20B	500483	7052497	2.0	113
SNAP23	505390	7053358	2.0	270
SNAP29	506563	7053378	1.5	295
Far-field				
SNAP04	509952	7054110	1.5	495
SNAP07	510816	7053351	2.0	462
SNAP08	511872	7053958	1.5	488
SNAP10	508801	7049847	1.5	479
Mid-field				
SNAP09	509851	7051660	2.0	458
SNAP11A	508729	7051700	2.0	470
Near-field				
SNAP03	507868	7053448	2.0	470
SNAP05	508376	7052958	2.0	468
SNAP06	509424	7052594	2.0	473
SNAP12	507753	7052652	1.5	476
SNAP26	506718	7052116	1.5	515
SNAP28	507021	7052790	1.5	460
Diffuser				
SNP 02-20d	507411	7052845	2.0	462
SNP 02-20e	507158	7052607	1.6	472
SNP 02-20f	507316	7052949	2.0	458

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 µS/cm

1,000 0 1,000

SCALE 1:33,000 METRES

NOTE

µS/cm = microSiemens per centimetre
m = metres

REFERENCE

Datum: NAD83 Projection: UTM Zone 12

PROJECT

DE BEERS
GROUP OF COMPANIES

TITLE

SURFACE CONDUCTIVITY IN SNAP LAKE, APRIL 2012

Golder Associates

PROJECT		13-1349-0001	FILE No.
DESIGN	KLN	03 Feb. 2012	SCALE AS SHOWN REV. 0
GIS	GU	09 Apr. 2013	
CHECK	LMB	19 Apr. 2013	FIGURE: 3-31
REVIEW	PMC	18 Apr. 2013	

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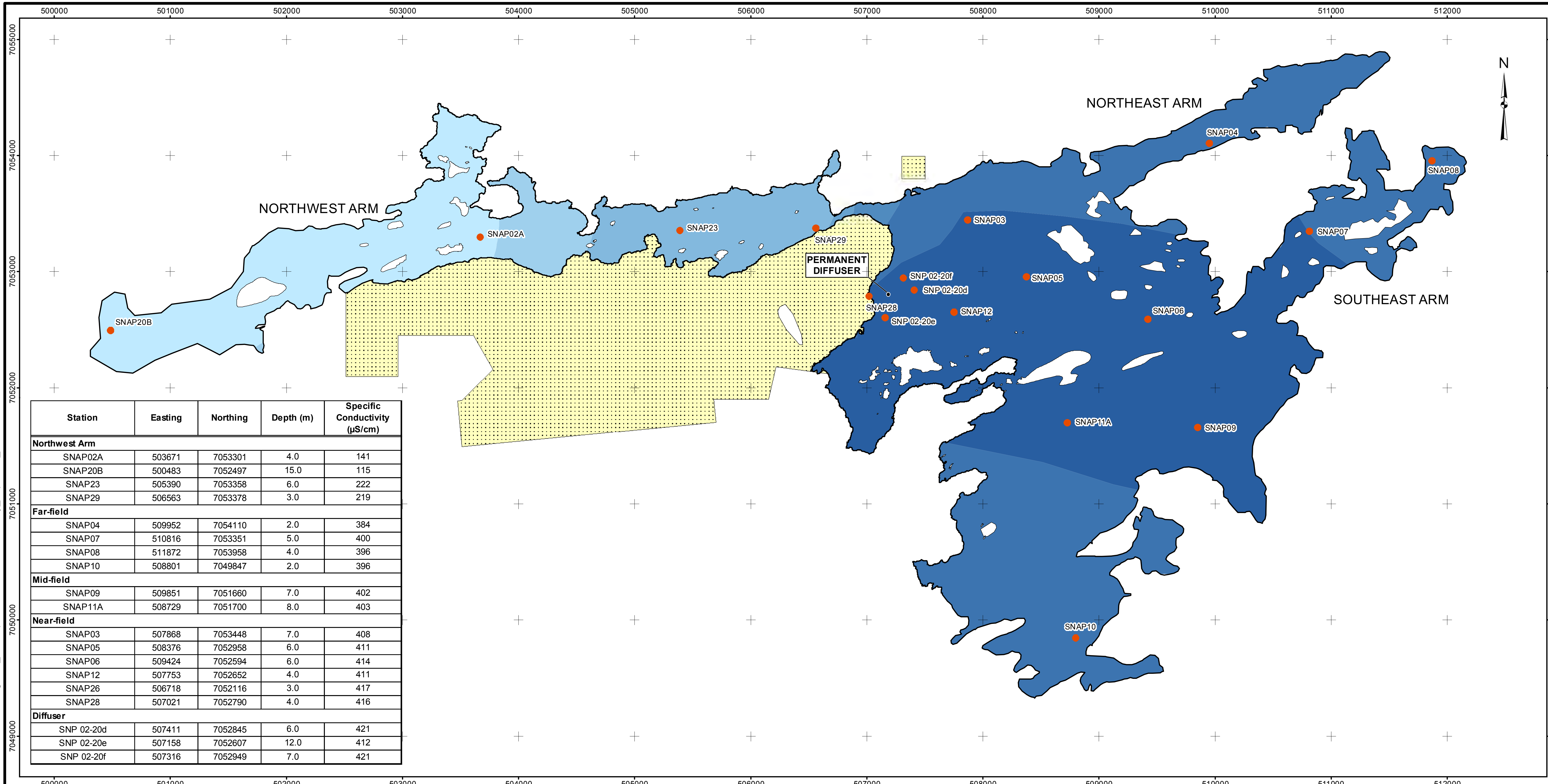

Station	Easting	Northing	Depth (m)	Specific Conductivity (µS/cm)
Northwest Arm				
SNAP02A	503671	7053301	4.0	141
SNAP20B	500483	7052497	15.0	115
SNAP23	505390	7053358	6.0	222
SNAP29	506563	7053378	3.0	219
Far-field				
SNAP04	509952	7054110	2.0	384
SNAP07	510816	7053351	5.0	400
SNAP08	511872	7053958	4.0	396
SNAP10	508801	7049847	2.0	396
Mid-field				
SNAP09	509851	7051660	7.0	402
SNAP11A	508729	7051700	8.0	403
Near-field				
SNAP03	507868	7053448	7.0	408
SNAP05	508376	7052958	6.0	411
SNAP06	509424	7052594	6.0	414
SNAP12	507753	7052652	4.0	411
SNAP26	506718	7052116	3.0	417
SNAP28	507021	7052790	4.0	416
Diffuser				
SNP 02-20d	507411	7052845	6.0	421
SNP 02-20e	507158	7052607	12.0	412
SNP 02-20f	507316	7052949	7.0	421

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 µS/cm

1,000 0 1,000
SCALE 1:33,000 METRES

NOTE

µS/cm = microSiemens per centimetre
m = metres

REFERENCE

Datum: NAD83 Projection: UTM Zone 12

PROJECT
DE BEERS
GROUP OF COMPANIES

TITLE
MID-DEPTH CONDUCTIVITY IN SNAP LAKE, SEPTEMBER 2012

Golder Associates

PROJECT	13-1349-0001		FILE No.
DESIGN	KLN	03 Feb. 2012	SCALE AS SHOWN REV. 0
GIS	GU	09 Apr. 2013	
CHECK	LMB	19 Apr. 2013	**FIGURE: 3-32**
REVIEW	PMC	18 Apr. 2013	

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Dissolved Oxygen

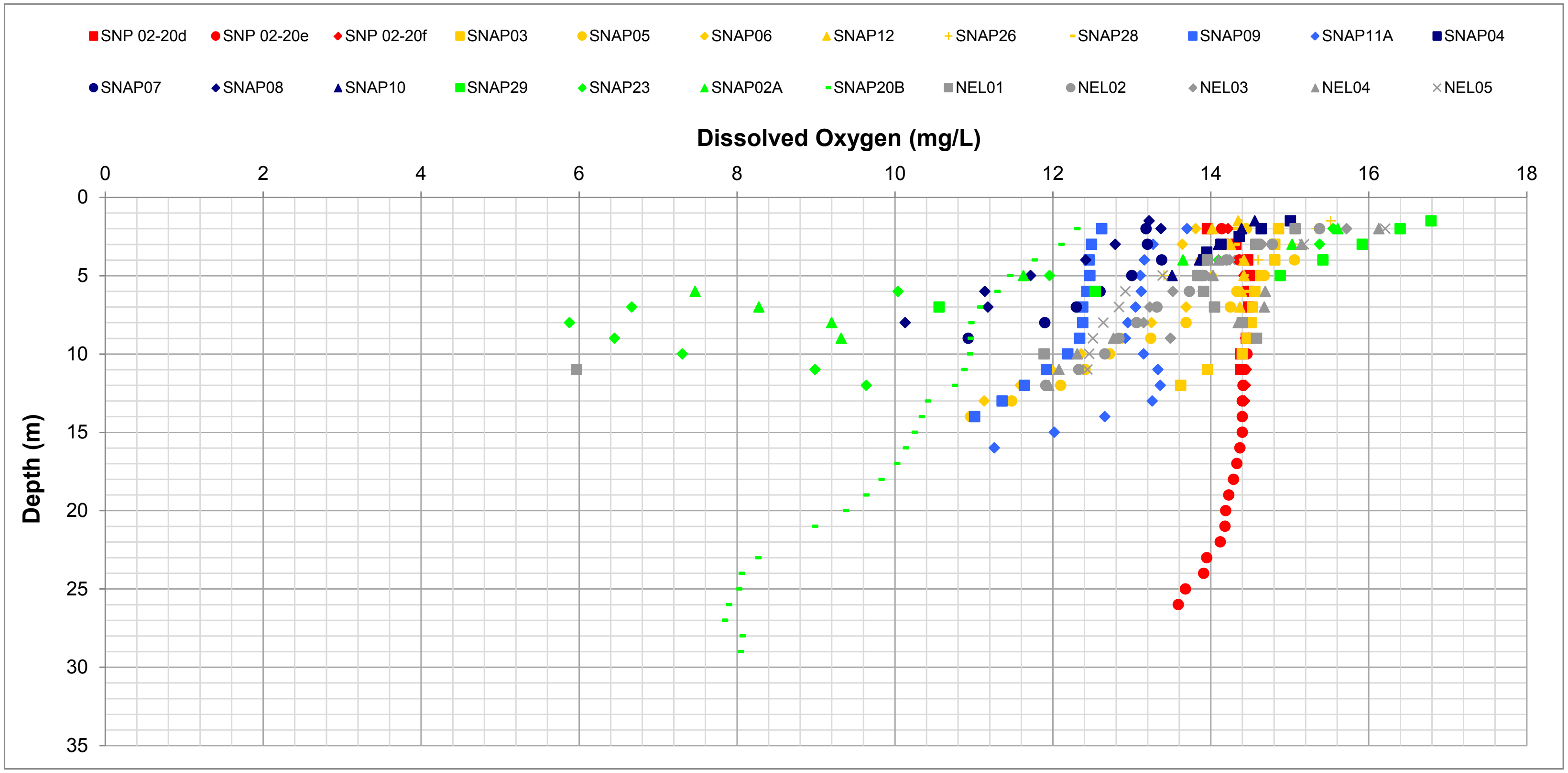
Concentrations of DO in Snap Lake varied by season and with water depth, ranging from 1.0 mg/L in January to 18.2 mg/L in July 2012; both measurements were from the northwest arm (Appendix 3B; Table 3B-1). At most monitoring locations, vertical gradients in DO occurred during ice-covered conditions, with lower DO concentrations evident near the bottom of the lake (Figure 3-33 and Figure 3-10, 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 subarctic lakes during winter conditions (Wetzel 2001). This vertical gradient did not occur at the shallower diffuser stations (SNP 02-20d and SNP 02-20f) during ice-covered conditions, where the DO concentrations were similar throughout the water column (Figures 3-33 and Figure 3-10 panels i and k). The treated effluent, which is well-oxygenated, may be increasing naturally low levels of DO at these shallower diffuser stations during ice-covered conditions.

Maximum DO concentrations during open-water conditions were lower than maximum ice-covered concentrations (Figure 3-34 and Figure 3-10 panels a to k), which is also consistent with natural DO variation. As the temperature of lake water decreases during the ice-covered season, the saturation point of DO increases, and therefore the water has a higher capacity for DO.

Vertical gradients in DO were not observed during open-water conditions in Snap Lake, with the exception of the deepest station in the northwest arm, SNAP20B, in July and September (Figure 3-34 and Figure 3-10 panel b) and the deepest diffuser station, SNP 02-20e, in the main basin of Snap Lake in July (Figure 3-10 panel j). Concentrations of DO increased in both deep stations in the lower 10 m of the water column in July (by up to 2 mg/L), likely related to the decrease in the water temperature at the same depth. In September, a decrease of about 4 mg/L between the surface and bottom DO concentration was observed at SNAP20B (Figure 3-33).

Similar to Snap Lake, vertical DO gradients occurred during 2012 ice-covered conditions in Northeast Lake, with lower DO concentrations near the bottom of the lake. Vertical gradients in DO were also observed during open-water conditions at the deep station in Northeast Lake (NEL06) in September (Figures 3-33 and 3-24, and Figure 3-10 panel l).

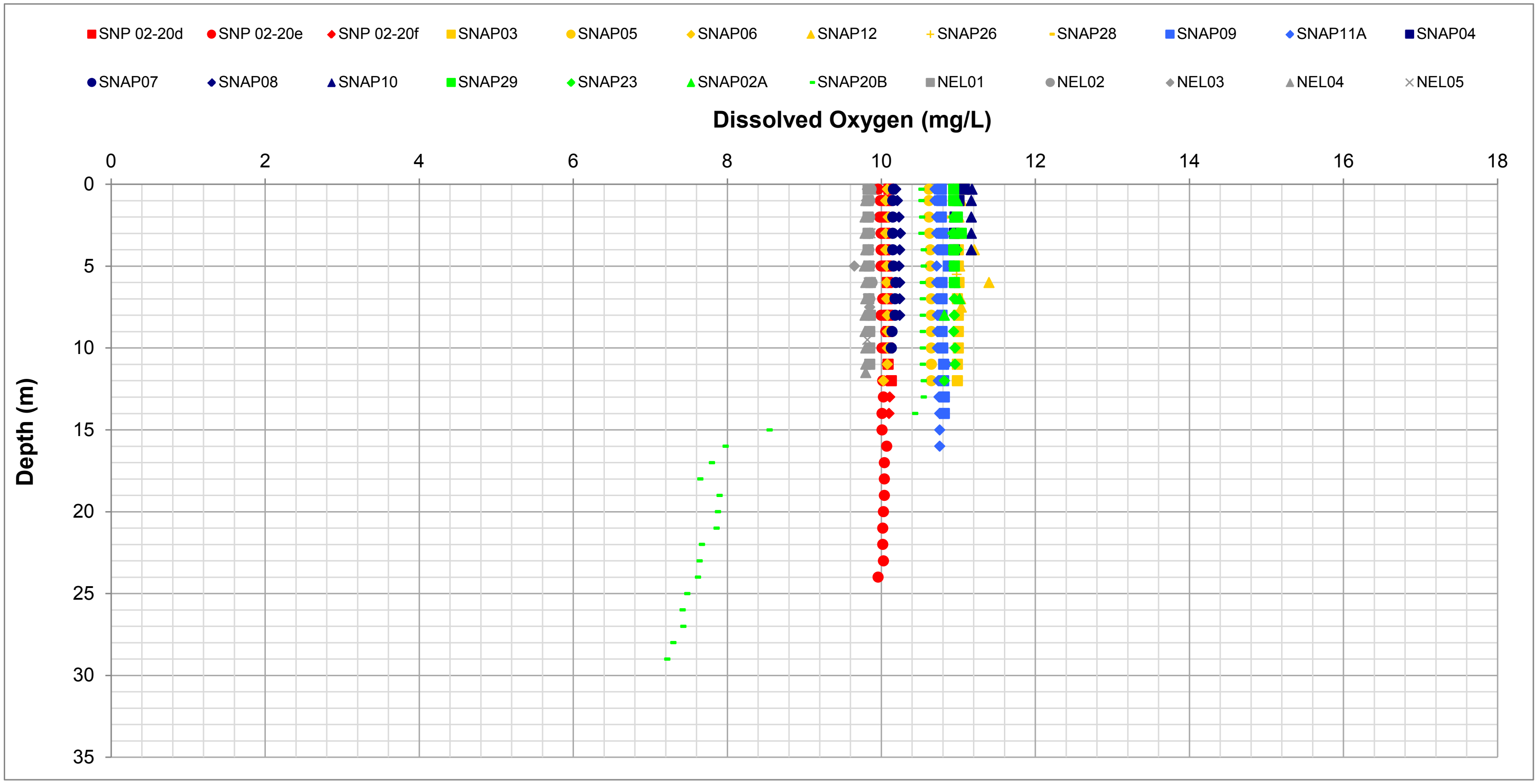
Figure 3-33 Dissolved Oxygen Profiles in Snap Lake and Northeast Lake, April 2012



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.

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

Figure 3-34 Dissolved Oxygen Profiles in Snap Lake and Northeast Lake, September 2012



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.

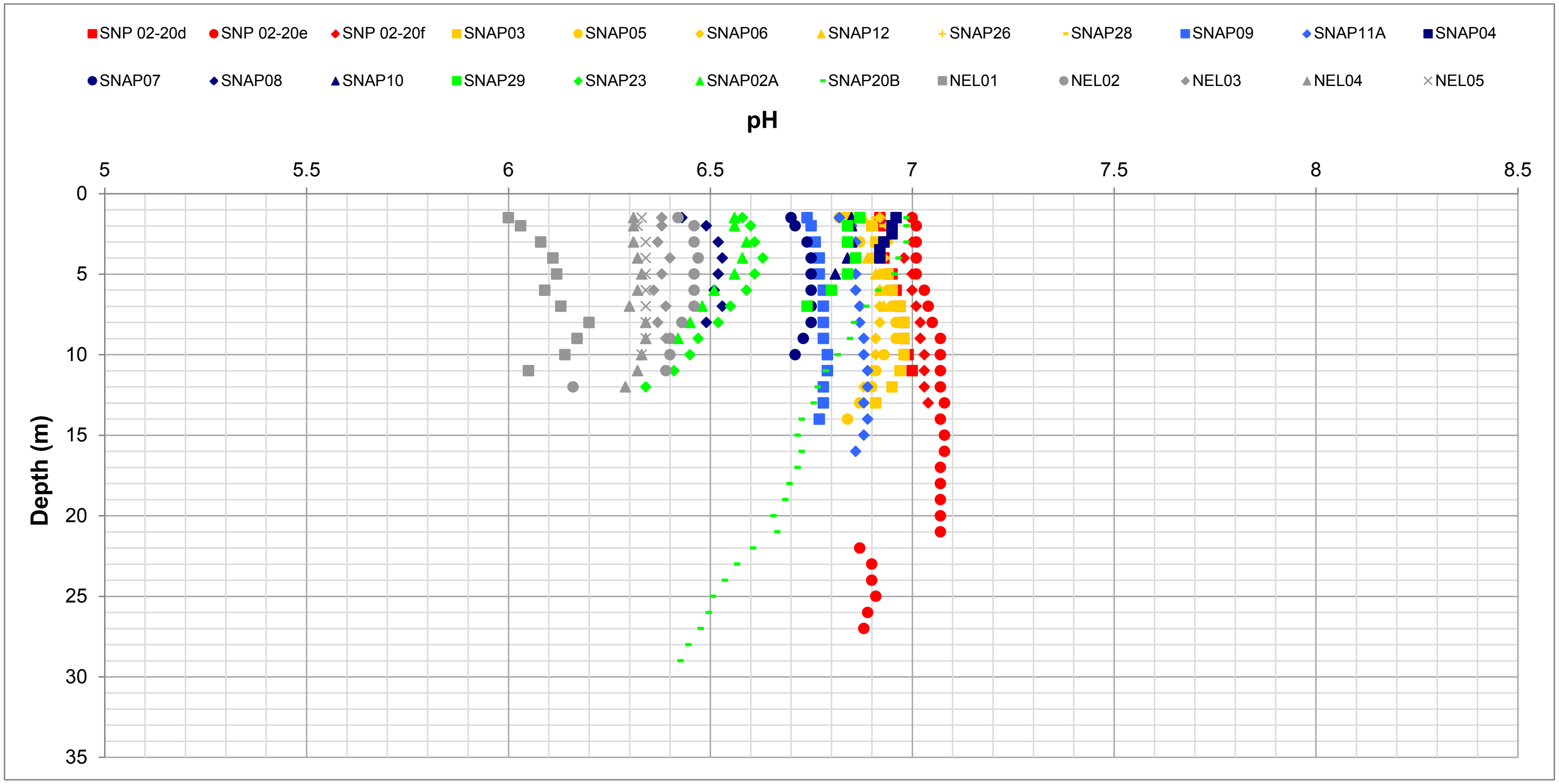
m = metre; mg/L = milligram per litre.

pH

In 2012, the range in Snap Lake field pH values was 5.6 to 7.5, which is a similar range to that measured in previous years (5.3 to 8.1) (De Beers 2005a, 2006, 2007a, 2008a, 2009, 2010, 2011a, 2012b). Treated effluent has elevated pH and alkalinity due to the high neutralization potential of kimberlite, and the elevated TDS and hardness. The pH of the treated effluent is adjusted and maintained during the treatment process. Vertical trends in pH at stations closest to the diffuser are similar to natural trends observed in Northeast Lake and the northwest arm of Snap Lake (Figures 3-35 to 3-38).

Slight decreases in pH values with increasing depth were observed at the majority of stations in Snap Lake during ice-covered conditions (Figures 3-35 and 3-36). This pattern can be attributed to an accumulation of carbon dioxide, primarily due to the influence of respiration processes in the deeper parts of the lake. Mixing of the water column during the open-water month of July 2012 resulted in more consistent pH values with depth at shallower stations (Figure 3-37); however, near bottom decreases in pH were observed at the deeper stations (SNP 2-20e and SNAP20B). This effect may be caused by deep water processes, such as respiration, sediment decomposition, and redox reactions. The vertical spatial patterns observed at SNP 02-20e and SNAP20B in July 2012 were consistent with the 2011 results (De Beers 2012b). The vertical spatial pattern observed at SNAP20B has been consistent since 2008 (De Beers 2009, 2010, 2011a). The pH values measured in September 2012 at SNAP20B decreased with depth similar to July 2012, although the pH values measured at SNP 02-20e increased with depth (Figure 3-38).

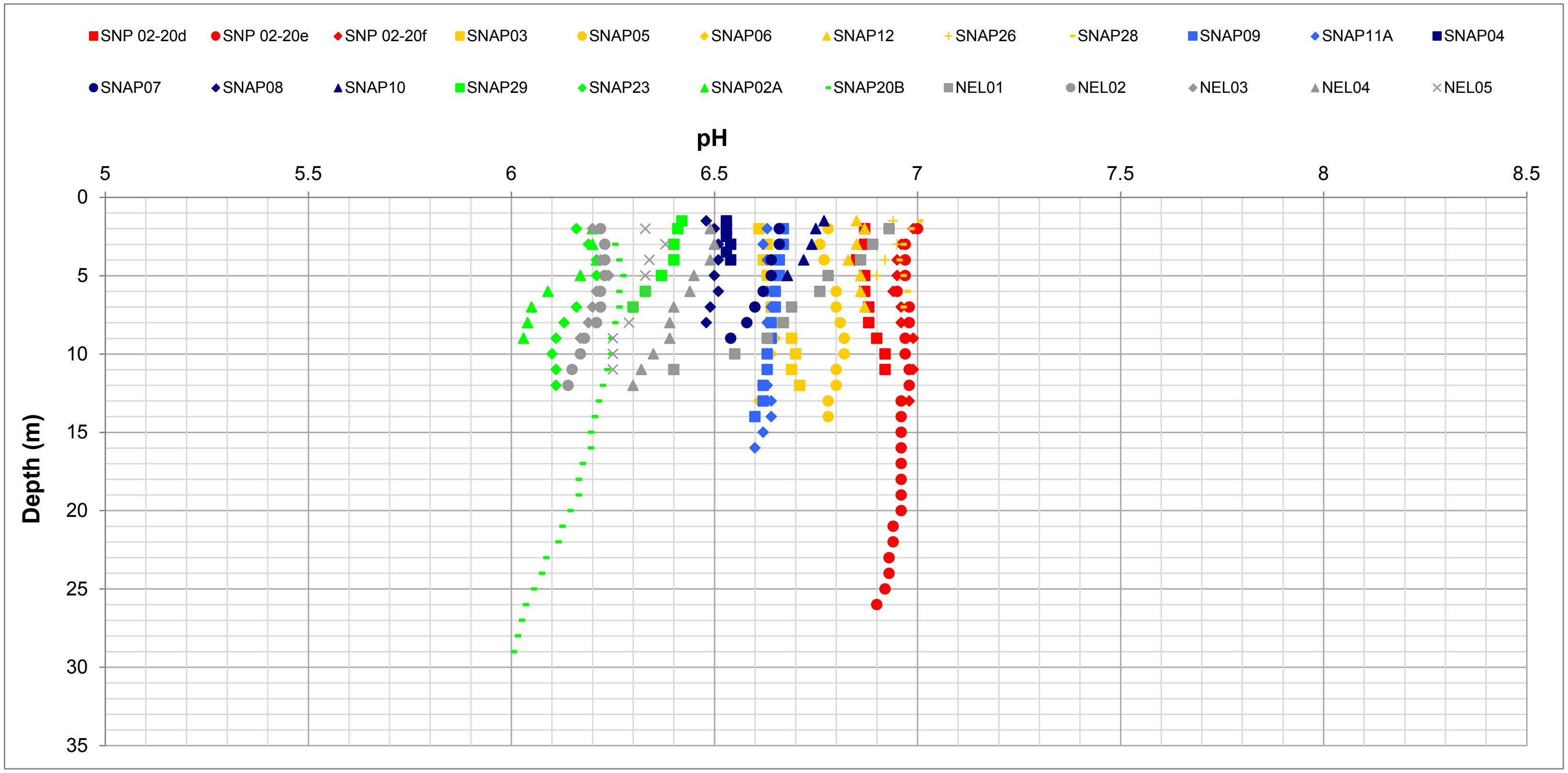
Figure 3-35 pH Profiles in Snap Lake and Northeast Lake, February 2012



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.

m = metre.

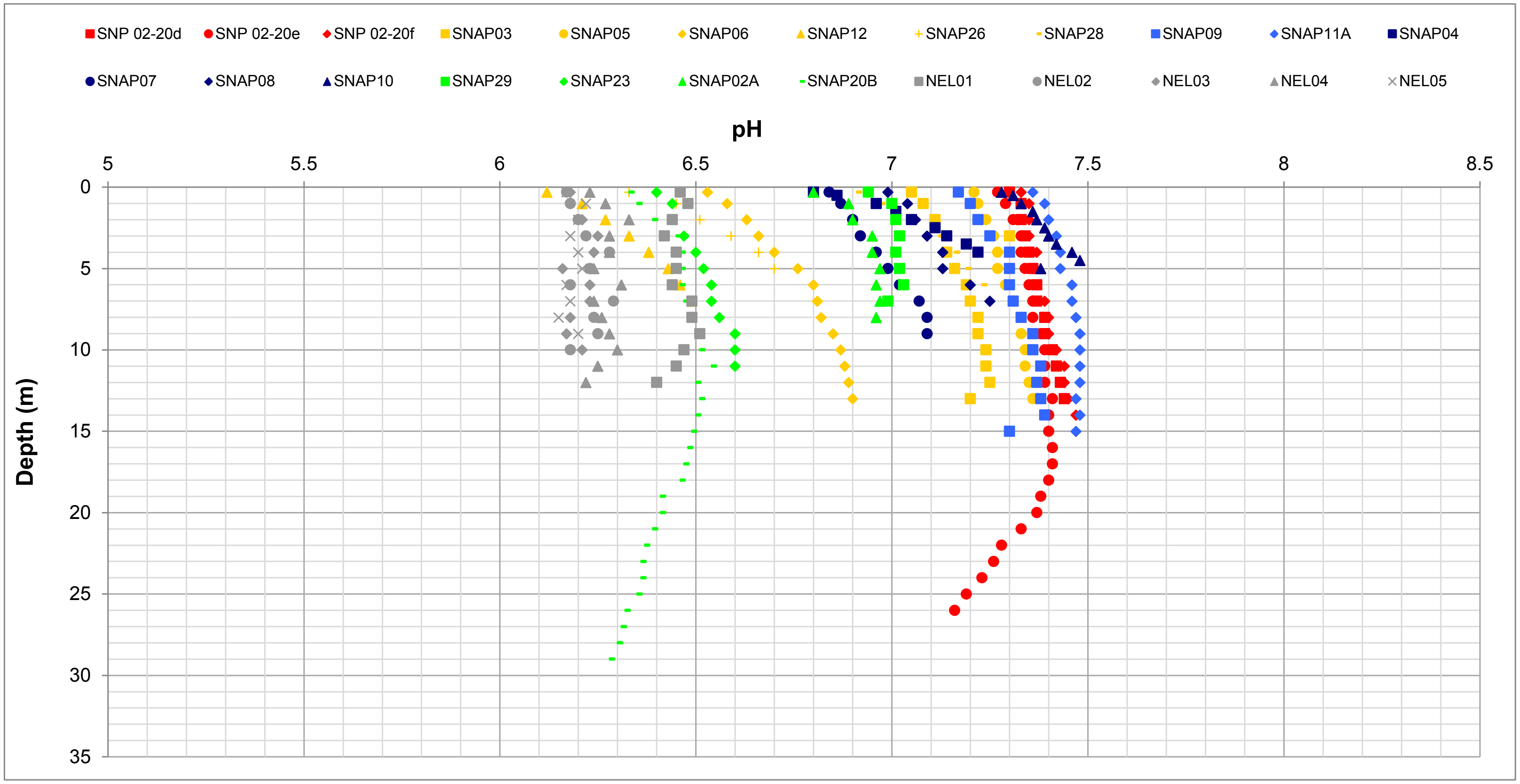
Figure 3-36 pH Profiles in Snap Lake and Northeast Lake, April 2012



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.

m = metre.

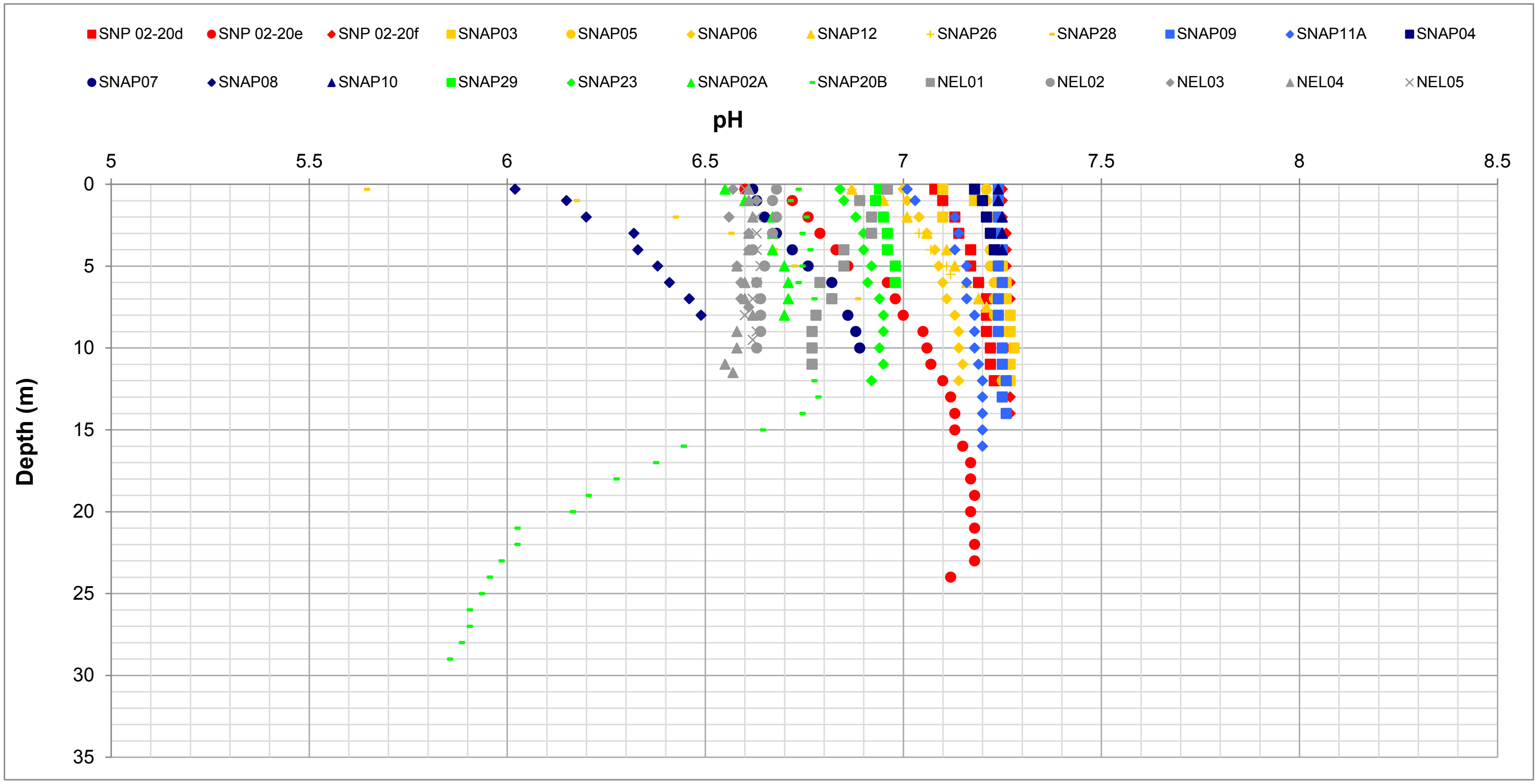
Figure 3-37 pH Profiles in Snap Lake and Northeast Lakes, July 2012



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.

m = metre.

Figure 3-38 pH Profiles in Snap Lake and Northeast Lake, September 2012



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.

m = metre.

Water Temperature

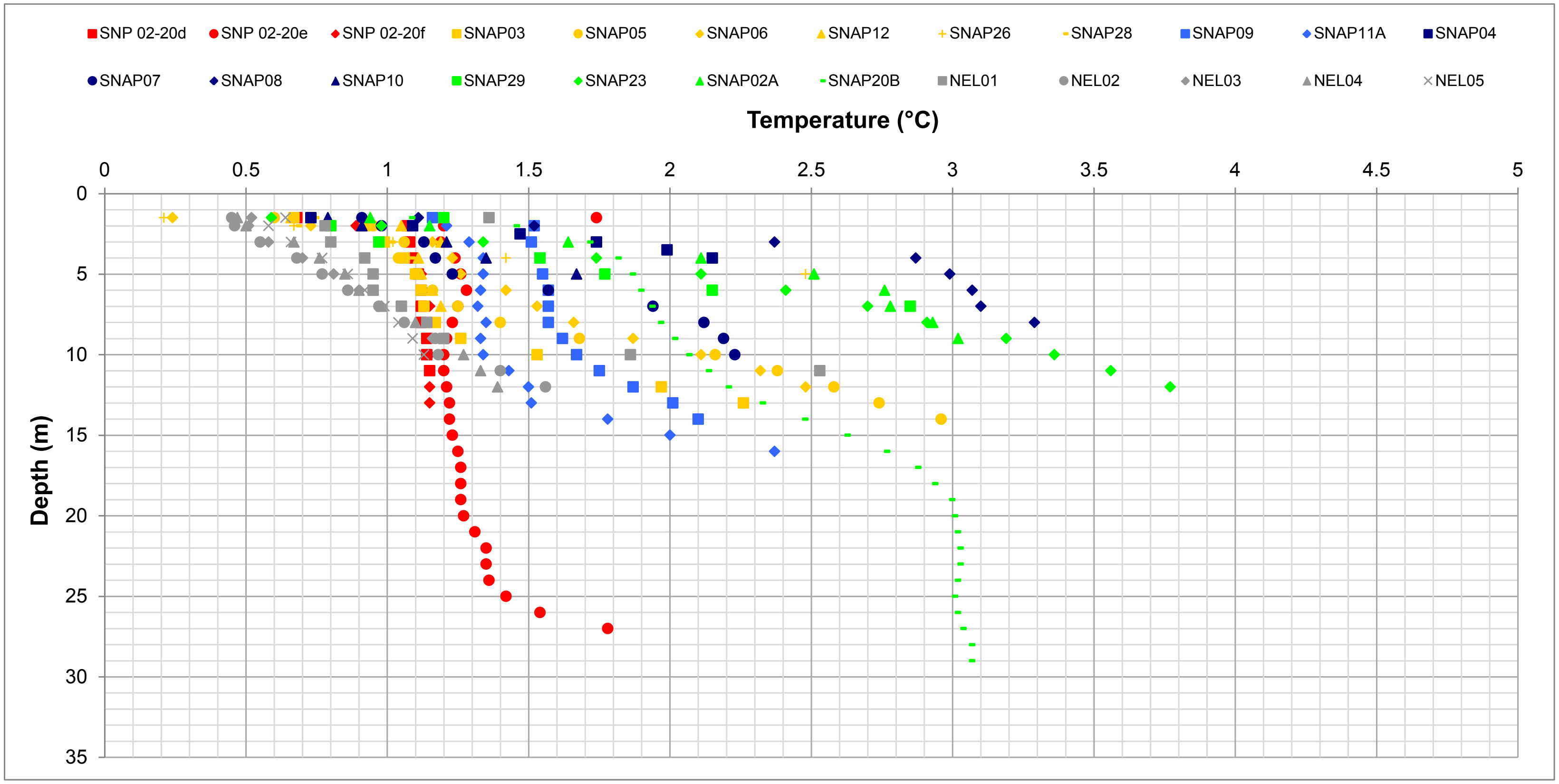
In 2012, surface water temperatures in Snap Lake varied from 0.2 degrees Celsius (°C) during ice-covered conditions in February (Figure 3-39) to 18.1°C during open-water conditions in July (Figure 3-41), similar to observations in previous programs. Consistent with 2006 to 2011, vertical temperature gradients were observed at:

- most stations during the ice-covered season (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 the ice-covered season, temperatures at all stations increased with depth although the temperature increase with depth at the diffuser stations was less prominent than at the other stations (Figures 3-39 to 3-42). During ice-covered conditions, the water column profiles at the diffuser and near-field stations were cooler than the water column profiles at the other locations in the lake. The temperature of the effluent in the WTP and TWTP (SNP 02-17B and SNP 02-17, respectively) in winter was approximately 5°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. After ice break-up, the shallow surface depths, including the euphotic zone of the lake warmed, so temperatures at all the stations decreased with depth (Figure 3-41). When the water column was mixed in September, the temperatures were relatively uniform through the water column, except at the deepest station in the northwest arm, SNAP20B. The temperature at SNAP20B remained low closer to the bottom of the lake (deeper than 15 m), indicating that the water was not completely mixed (i.e., a thermocline existed) at this deep location.

Temperature in Northeast Lake increased with depth during the ice-covered season (Figures 3-39 and 3-40), decreased with depth during the early open-water season in July (Figure 3-41), and was uniform throughout the water column in the late open-water season in September (Figure 3-42). During September, temperatures in Northeast Lake were higher than temperatures observed at Snap Lake stations, with the exception of three stations in the near-field area (SNAP03, SNAP12, and SNAP28; 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), with the exception of temperatures measured in 2010 (De Beers 2011a).

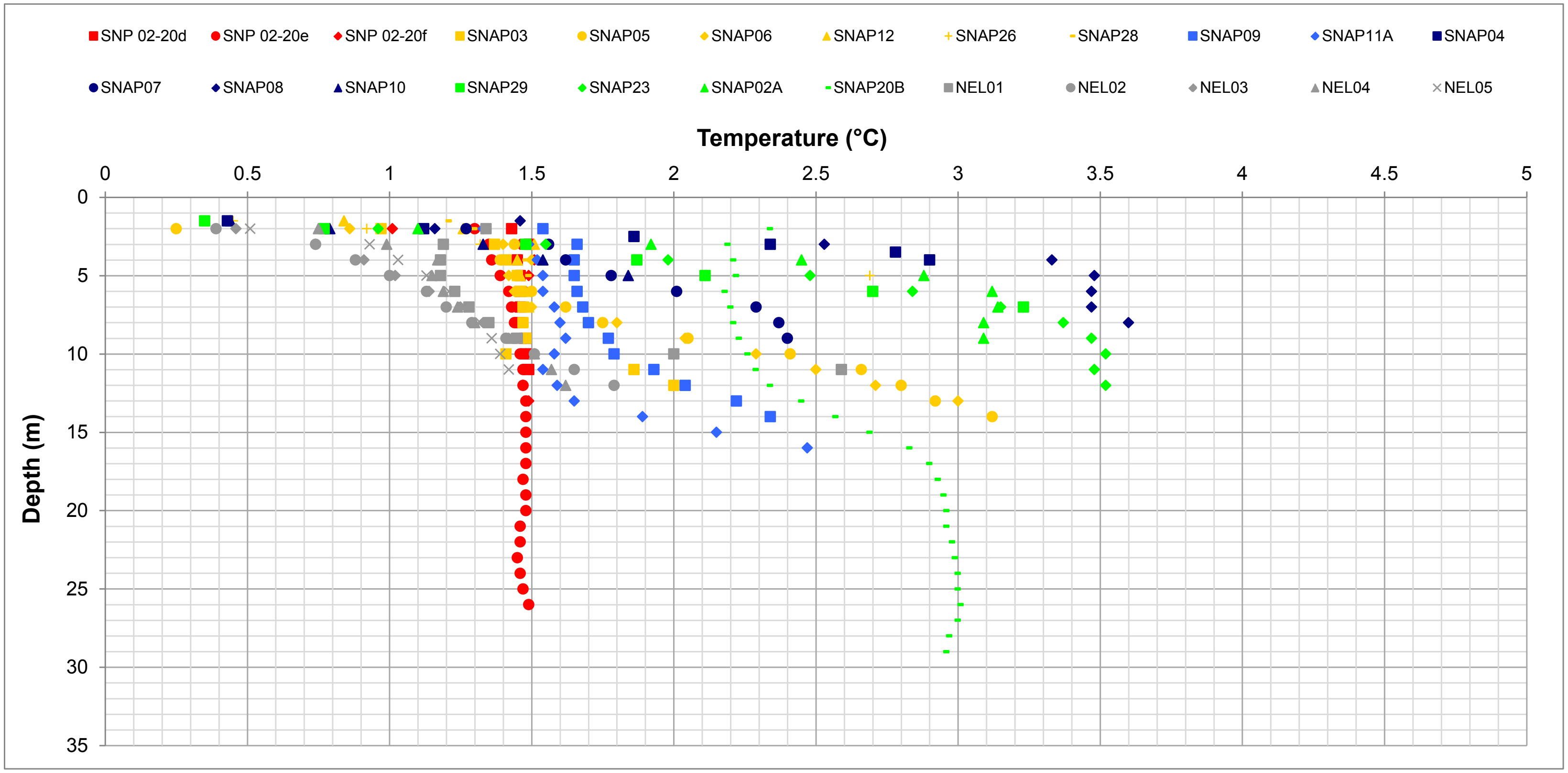
Figure 3-39 Water Temperature Profiles in Snap Lake and Northeast Lake, February 2012



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.

m = metre; °C = degrees Celsius.

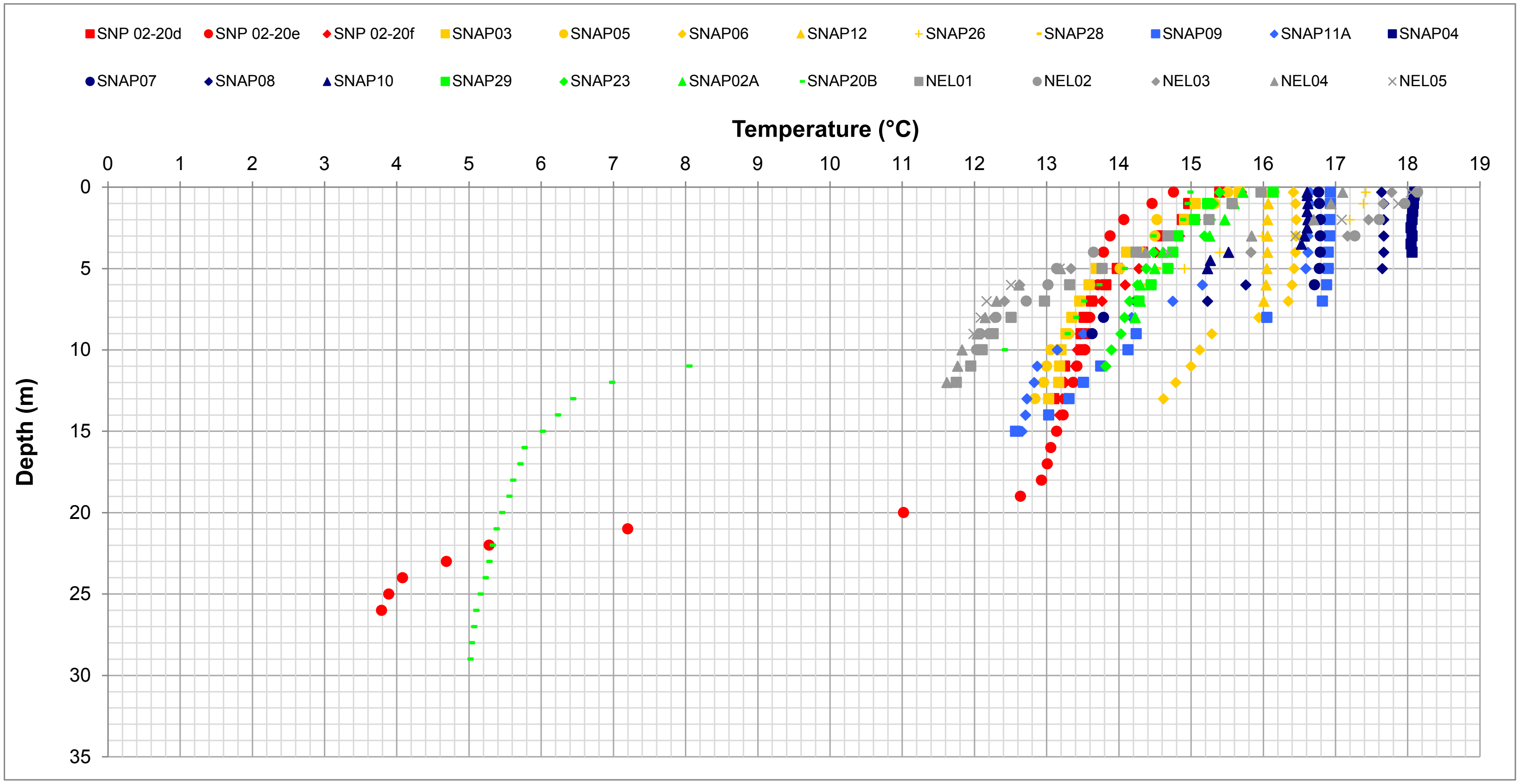
Figure 3-40 Water Temperature Profiles in Snap Lake and Northeast Lake, April 2012



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.

m = metre; °C = degrees Celsius.

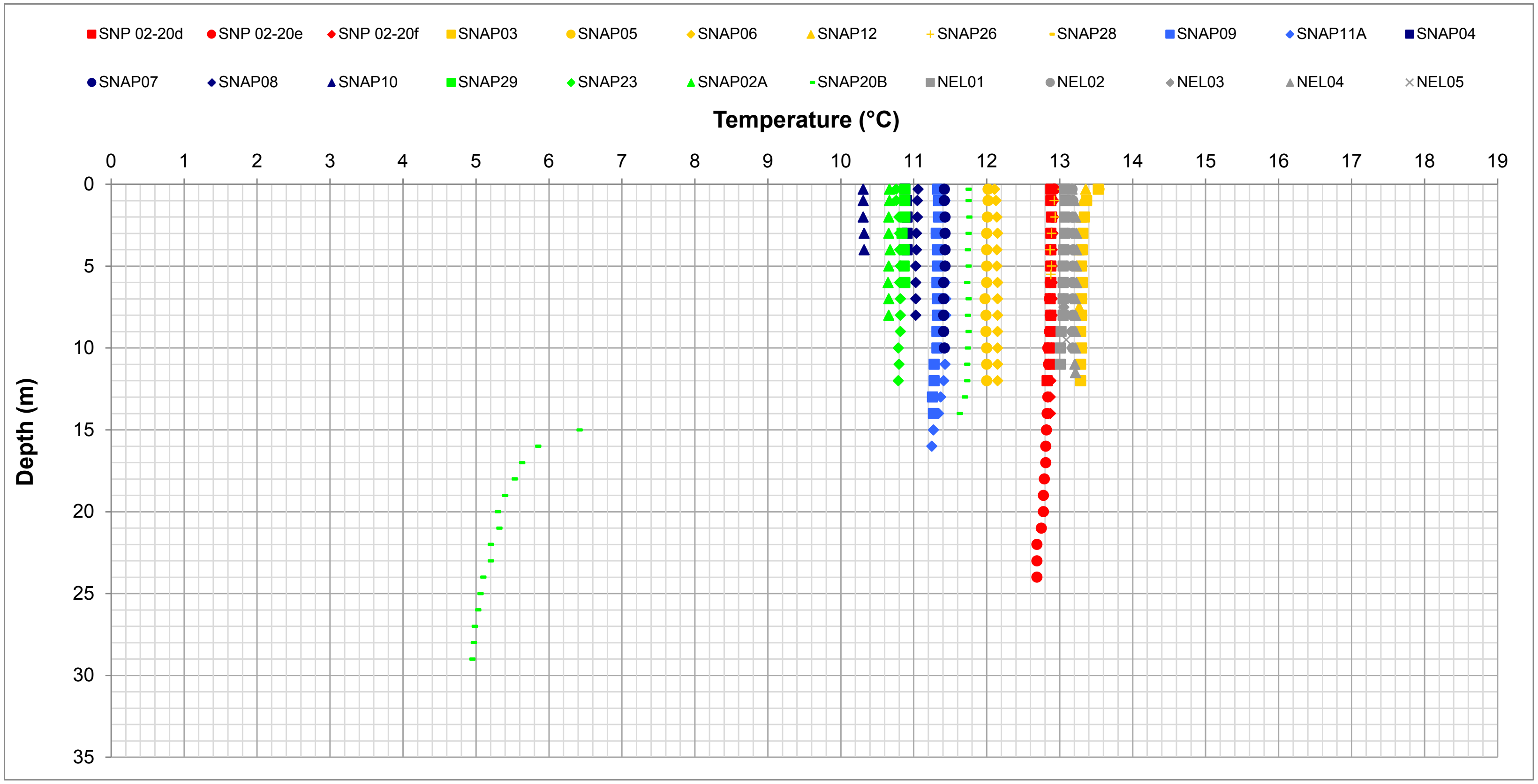
Figure 3-41 Water Temperature Profiles in Snap Lake and Northeast Lake, July 2012



Note: Grey symbols represent Northeast Lake stations; green symbols represent the northwest arm stations; light blue symbols represent the mid-field stations; dark blue colored symbols represent the far-field stations; yellow symbols represent the near-field stations; and, red symbols represent the diffuser stations.

m = metre; °C = degrees Celsius.

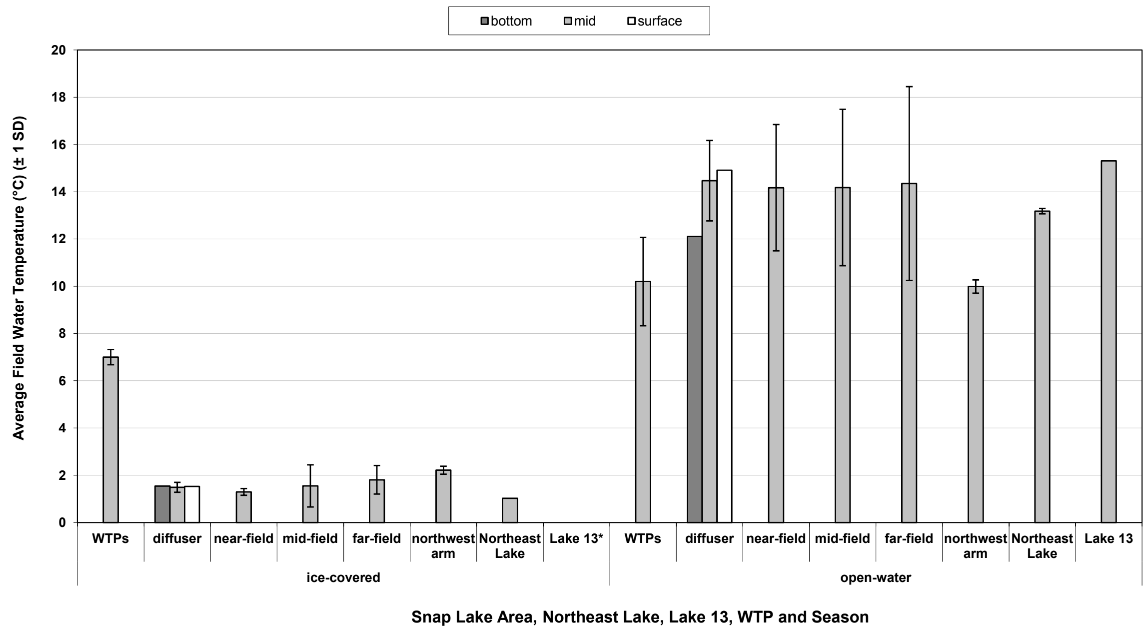
Figure 3-42 Water Temperature Profiles in Snap Lake and Northeast Lake, September 2012



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.

m = metre; °C = degrees Celsius.

Figure 3-43 Average Water Temperature in Snap Lake, Northeast Lake, Lake 13, and Water Treatment Plants, 2012



* Water samples were not collected from Lake 13 during the ice-covered season.

WTPs = temporary and permanent water treatment plants (SNP 02-17 and SNP 02-17B); SD = standard deviation; °C = degrees Celsius.

Total Dissolved Solids and Ions

Plots of TDS and major ions show three main spatial and seasonal patterns in 2012 (Figures 3-44 to 3-47):

- Average concentrations of TDS and major ions during open-water and ice-covered conditions were generally similar throughout the main basin of Snap Lake, but the main basin concentrations were notably higher than the northwest arm concentrations. Average concentrations in Northeast Lake and Lake 13 were also plotted for comparison and were substantially lower than average concentrations in any area of Snap Lake.
- Surface concentrations of TDS and major ions near the diffuser were typically lower compared to mid-depth and bottom during ice-covered conditions; during open-water conditions, average concentrations near the diffuser were consistent with depth.
- Average TDS and major ions concentrations were higher during ice-covered conditions when compared to open-water conditions.

The lack of spatial variability in the main basin of Snap Lake is consistent with trends in decreasing spatial variations in the main basin of Snap Lake over time (De Beers 2012b). The difference between TDS concentrations at the diffuser stations and the far-field stations was approximately 19 mg/L during the ice-covered and open-water seasons in 2012 (Figure 3-44).

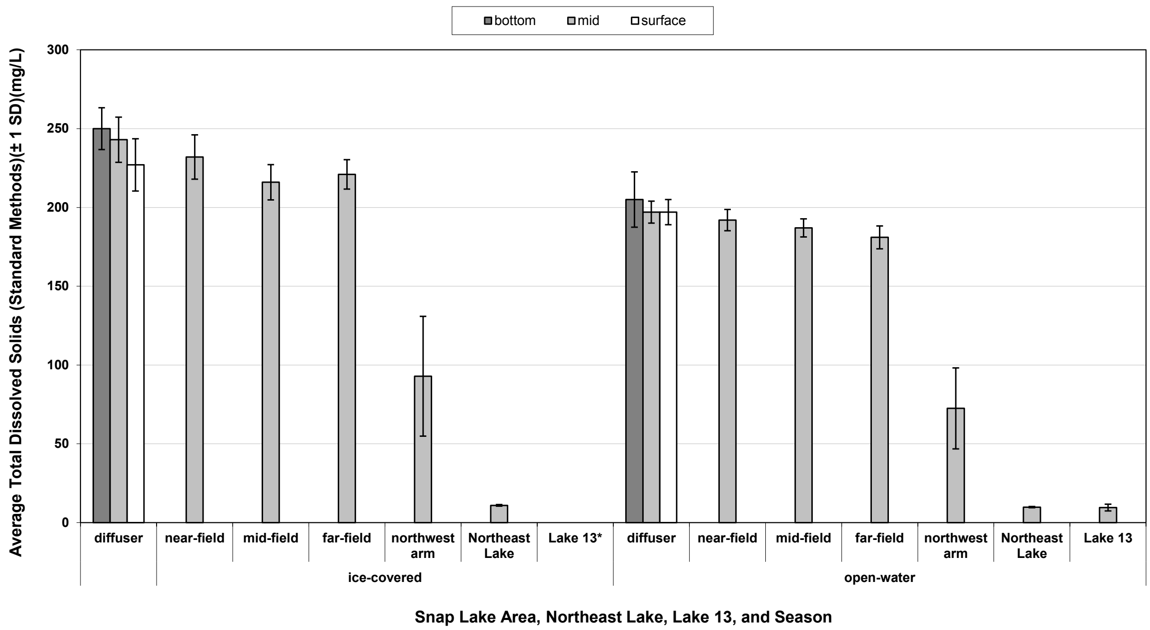
These differences represent relative percent differences between the diffuser and the far-field stations of 8% and 10% during ice-cover and open-water conditions, respectively. These are small percentage differences, indicating that Snap Lake is a well-mixed system. Concentrations in the main basin remain above northwest arm concentrations due to the limited hydraulic connectivity between the main basin and northwest arm. These patterns are consistent with the field measurements of conductivity, as discussed in Section 3.4.5.1.

Bottom concentrations of TDS and major ions at the diffuser stations were slightly higher than the mid-depth concentrations during ice-covered conditions (Figures 3-44 to 3-47). During open-water conditions, differences in average concentrations of TDS and major ions from different depths near the diffuser are not discernible. This pattern is consistent with the conductivity profiles, which indicated that the plume may no longer be sinking to the bottom of the lake during ice-covered conditions, 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 forces, such as wind-driven currents, have a greater influence on the plume compared to the effects of relative water densities.

Maximum average concentrations of major ions in Snap Lake typically occurred during ice-covered 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 major ions are 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.

Concentrations of TDS and major ions were similar between Northeast Lake and Lake 13 during open-water conditions.

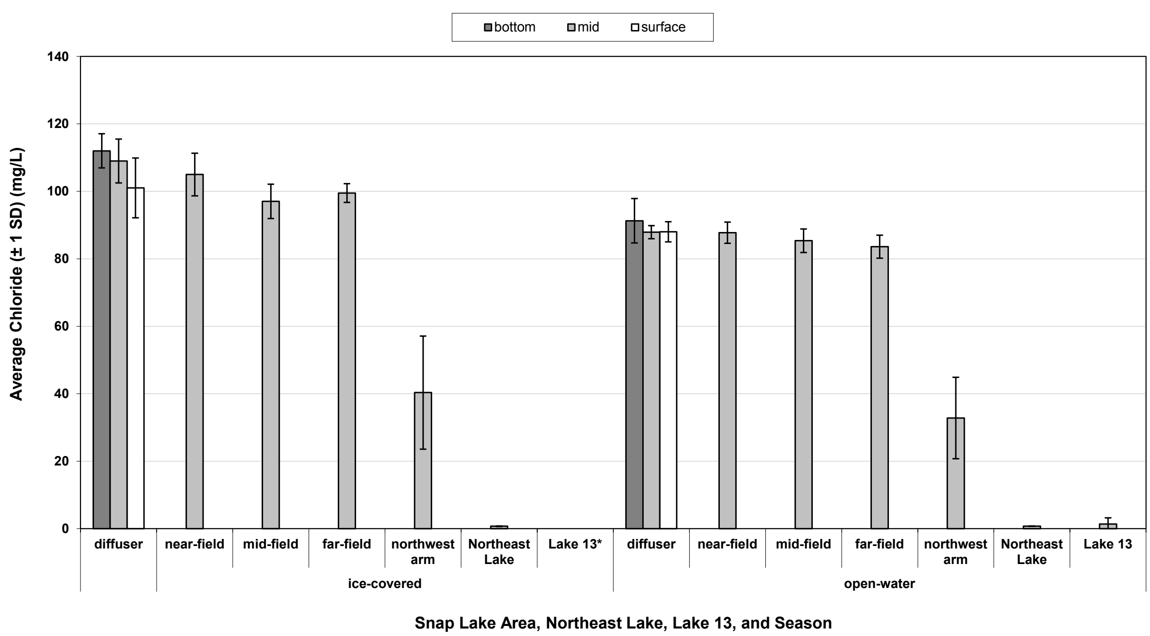
Figure 3-44 Average Calculated Total Dissolved Solids Concentrations (± 1 SD) in Snap Lake, Northeast Lake, and Lake 13, 2012



* Water samples were not collected from Lake 13 during the ice-covered season.

SD = standard deviation; mg/L = milligrams per litre.

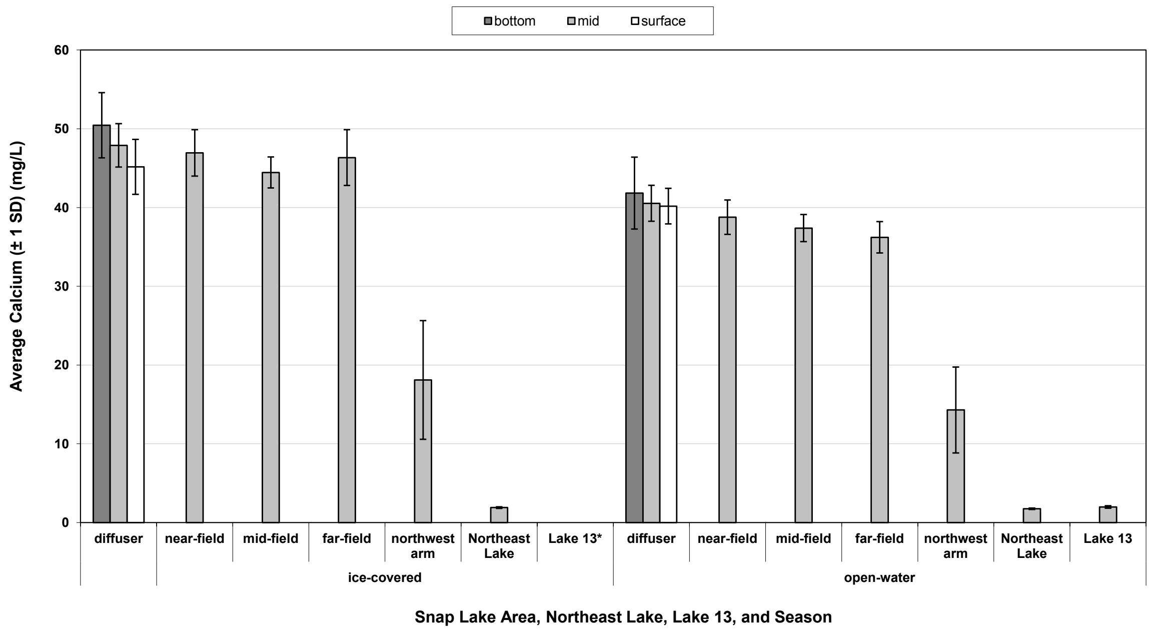
Figure 3-45 Average Chloride Concentrations (± 1 SD) in Snap Lake, Northeast Lake, and Lake 13, 2012



* Water samples were not collected from Lake 13 during the ice-covered season.

SD = standard deviation; mg/L = milligrams per litre.

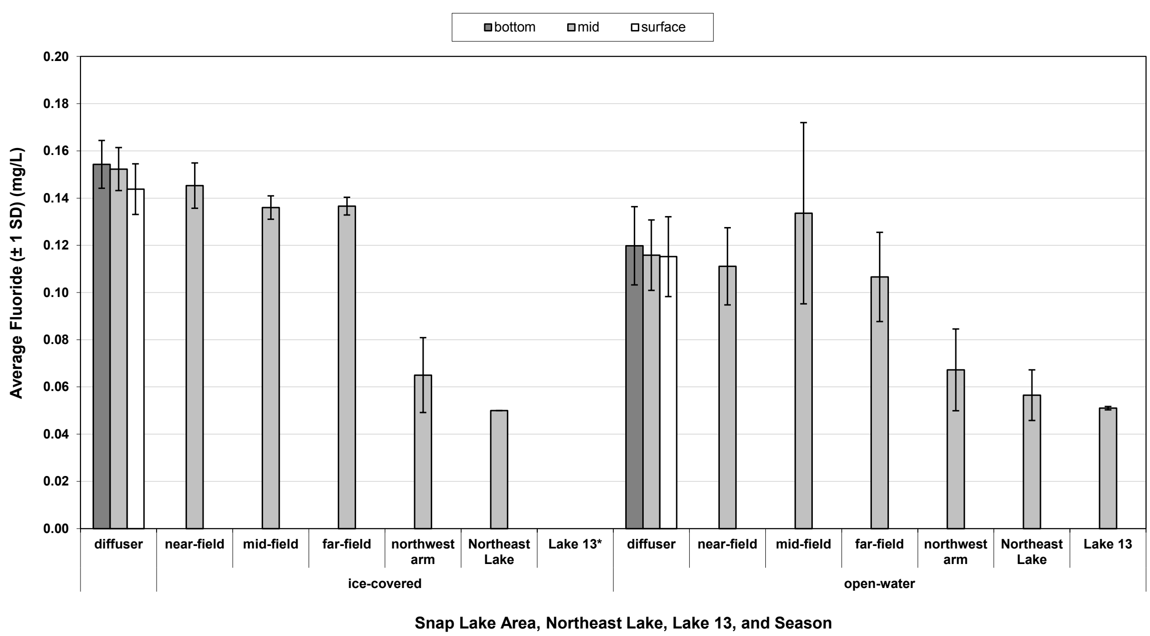
Figure 3-46 Average Calcium Concentrations (± 1 SD) in Snap Lake, Northeast Lake, and Lake 13, 2012



* Water samples were not collected from Lake 13 during the ice-covered season.

SD = standard deviation; mg/L = milligrams per litre.

Figure 3-47 Average Fluoride Concentrations (± 1 SD) in Snap Lake, Northeast Lake, and Lake 13, 2012



* Water samples were not collected from Lake 13 during the ice-covered season.

SD = standard deviation; mg/L = milligrams per litre.

Nutrients

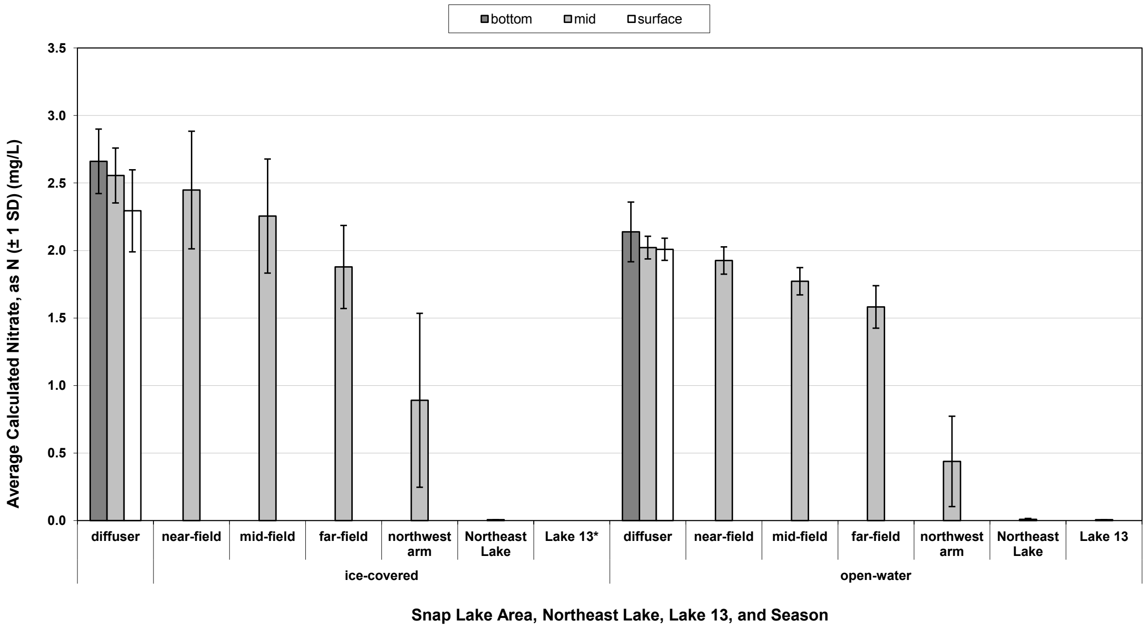
Concentrations of nitrate, nitrite, and ammonia in Snap Lake decreased with increasing distance from the diffuser (Figures 3-48 to 3-50). Higher concentrations of nutrients are expected in areas close to the diffuser influenced by the WTP discharge because the treated effluent contains elevated concentrations of both nitrogen and phosphorus (Appendix 3G).

Seasonal differences occurred in ammonia and nitrate concentrations in Snap Lake in 2012; average concentrations of ammonia and nitrate were higher during ice-covered conditions compared to open-water conditions (Figures 3-49 and 3-50). The decrease in ammonia and nitrate concentrations during open-water conditions may be due to assimilation by phytoplankton (ammonia would also readily nitrify (oxidize) to nitrate 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.

Clear spatial patterns in TP were not evident (Figure 3-51), consistent with previous years (De Beers 2010, 2011a, 2012b). The lack of a distinct spatial pattern in phosphorus may be related to the high variability in the low-level phosphorus concentrations in Snap Lake or rapid uptake of phosphorus by phytoplankton (Section 3.4.4), which reside at shallower depths within the euphotic zone (Section 12.4) than the mid-depth sampling depth, or a combination of both.

Concentrations of nutrients were typically similar between Northeast Lake and Lake 13, with the exception of TP; TP concentrations were higher in Lake 13.

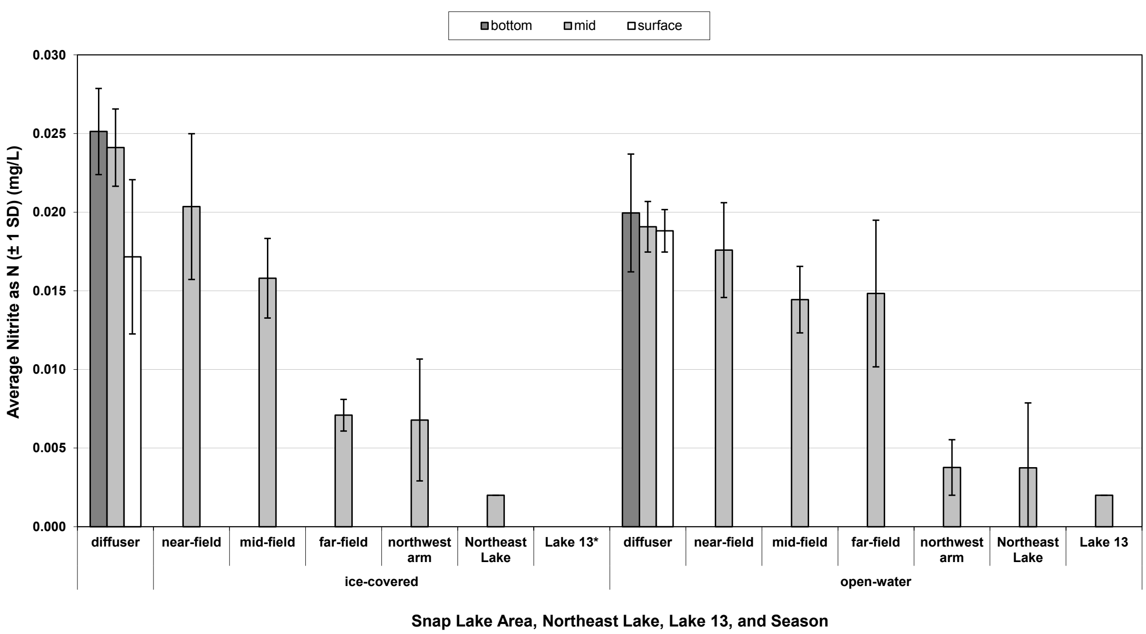
Figure 3-48 Average Calculated Nitrate Concentrations (± 1 SD) in Snap Lake, Northeast Lake, and Lake 13, 2012



* Water samples were not collected from Lake 13 during the ice-covered season.

SD = standard deviation; N = nitrogen; mg/L = milligrams per litre.

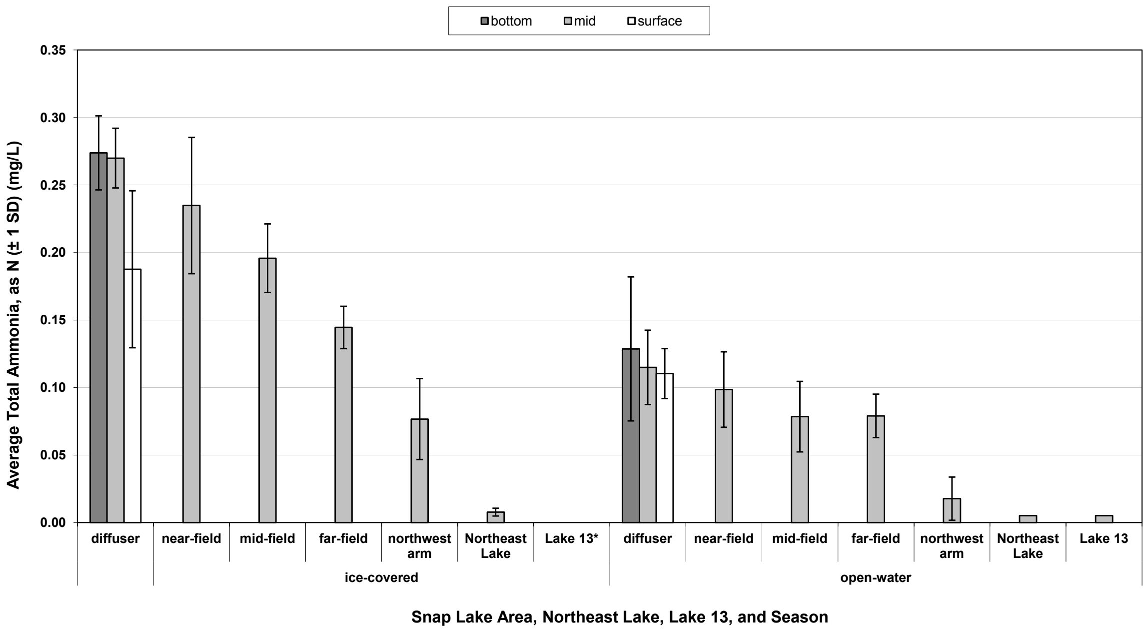
Figure 3-49 Average Nitrite Concentrations (± 1 SD) in Snap Lake, Northeast Lake, and Lake 13, 2012



* Water samples were not collected from Lake 13 during the ice-covered season.

SD = standard deviation; N = nitrogen; mg/L = milligrams per litre.

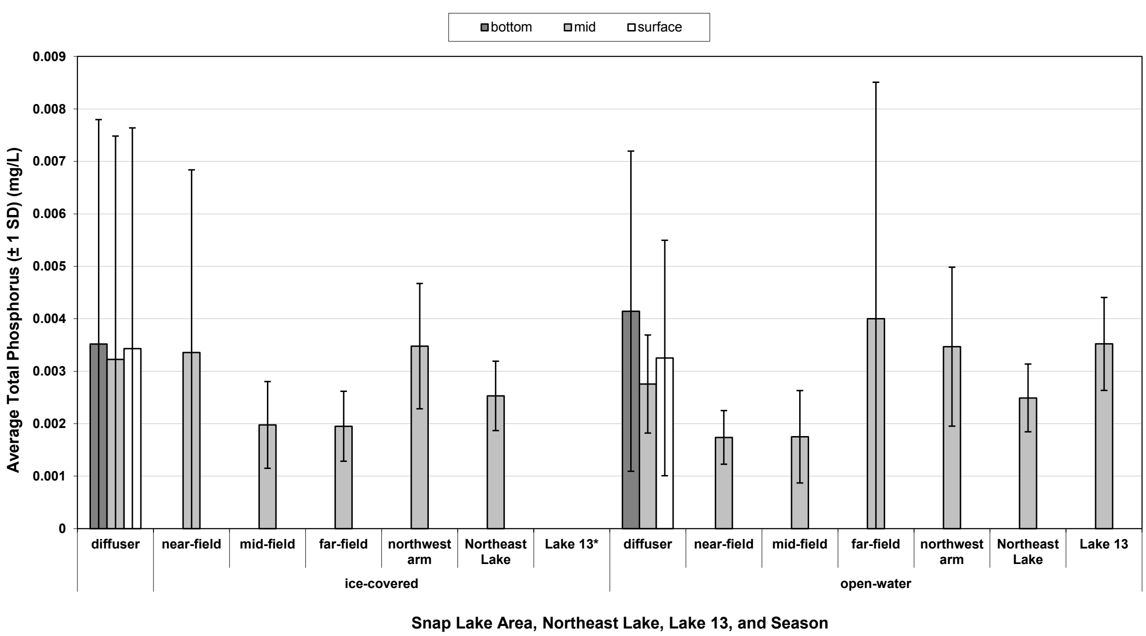
Figure 3-50 Average Total Ammonia Concentrations (± 1 SD) in Snap Lake, Northeast Lake, and Lake 13, 2012



* Water samples were not collected from Lake 13 during the ice-covered season.

SD = standard deviation; N = nitrogen; mg/L = milligrams per litre.

Figure 3-51 Average Total Phosphorus Concentrations (± 1 SD) in Snap Lake, Northeast Lake, and Lake 13, 2012



* Water samples were not collected from Lake 13 during the ice-covered season.

SD = standard deviation; mg/L = milligrams per litre.

Metals

Spatial and seasonal plots for several representative metals are presented in Figures 3-51 to 3-58; plots for the other metals are presented in Appendix 3G. Metals with strong positive correlations to conductivity (e.g., boron, barium, lithium, molybdenum, nickel, rubidium, strontium, uranium; Section 3.4.4), demonstrated 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., arsenic, cadmium, chromium, copper, iron, lead, manganese, mercury, titanium; Table 3-18). Concentrations of the metals were typically higher during ice-covered conditions compared to open-water conditions, with some exceptions as outlined in Table 3-18.

Table 3-18 Summary of Spatial and Seasonal Trends for Total Metals Measured in Snap Lake, 2012

Spatial/Seasonal Pattern	Metals that Apply
Spatial Pattern[(a)]	
Average concentrations were clearly higher in the main basin compared to the northwest arm (Examples: total barium and total molybdenum in Figures 3-52 and 3-53)	Boron, barium, lithium, molybdenum, nickel, rubidium, strontium, and uranium [(b)] • Boron, barium, lithium, rubidium, and strontium concentrations were similar across the main basin • A gradient was observed for molybdenum, nickel, and uranium, with highest concentrations measured near the diffuser and lowest concentrations in the far-field
Average concentrations were higher in the northwest arm compared to the main basin (Example: total manganese in Figure 3-54)	Aluminum (open-water), iron, and manganese[(c)]
Average concentrations were elevated near the diffuser, but much lower throughout rest of lake (Example: total antimony in Figure 3-55)	Antimony, cadmium, cobalt, titanium[(c)]
No clear spatial pattern (Example total arsenic in Figure 3-56)	Arsenic, chromium, copper, lead, mercury, and zinc[(c)] Beryllium, bismuth, cesium, hexavalent chromium, selenium, silver, thallium, vanadium all at or near the DL in 2012
Seasonal Pattern	
Average lake concentrations higher under ice-covered conditions (Example: total strontium in Figure 3-57)	Antimony, barium, boron, cadmium, chromium, cobalt, lithium, molybdenum, nickel, rubidium, strontium, uranium[(a)]
Average lake concentrations higher under open-water conditions (Example: total iron in Figure 3-58)	Aluminum, iron, lead, zinc
No clear seasonal pattern (Example: total copper in Figure 3-59)	Arsenic, copper, mercury, titanium Beryllium, bismuth, cesium, hexavalent chromium, manganese, selenium, silver, thallium, vanadium were all at or near the DL in 2012.

(a) Examples of total metals representing the spatial and seasonal trends, or lack of trend, are shown in Figures 3-52 to 3-59. The remaining metals are presented in Appendix 3G.

(b) Metals in the list were strongly correlated with conductivity (Section 3.4.4)

(c) Metals in the list were weakly correlated with conductivity (Section 3.4.3)

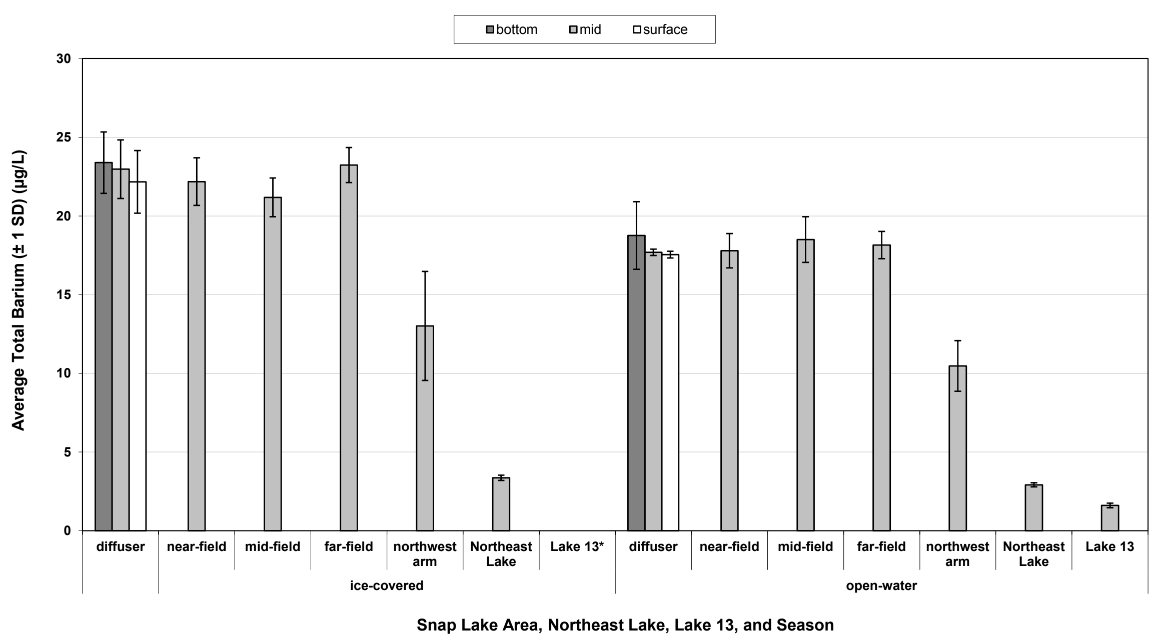
Manganese and iron concentrations were elevated at the deepest station in the northwest arm (SNAP20B) during ice-covered and open-water conditions, which contributed to the elevated average concentration for the northwest arm (Figures 3-54 and 3-58). The source of the elevated manganese and iron in the northwest arm is likely related to:

- Lower dissolved oxygen in the northwest arm (Section 3.4.3; Figure 3-10, panels a and b), and the reduction of manganese to the more soluble form under such conditions.
- Inflows from Stream S27 during open-water conditions. Iron concentrations in Stream S27 ranged from 176 to 282 mg/L during spring freshet, much higher than measured in the northwest arm (i.e., 4.2 to 31 mg/L)(Section 3.4.6).

Antimony, cadmium, cobalt, and titanium concentrations were higher at the diffuser stations during ice-covered conditions (Figure 3-54; Appendix 3G). For antimony and cadmium, the elevated concentrations were mainly at the surface of the water column. Total antimony concentrations are discussed further in Appendix 3A and Section 3.4.7.

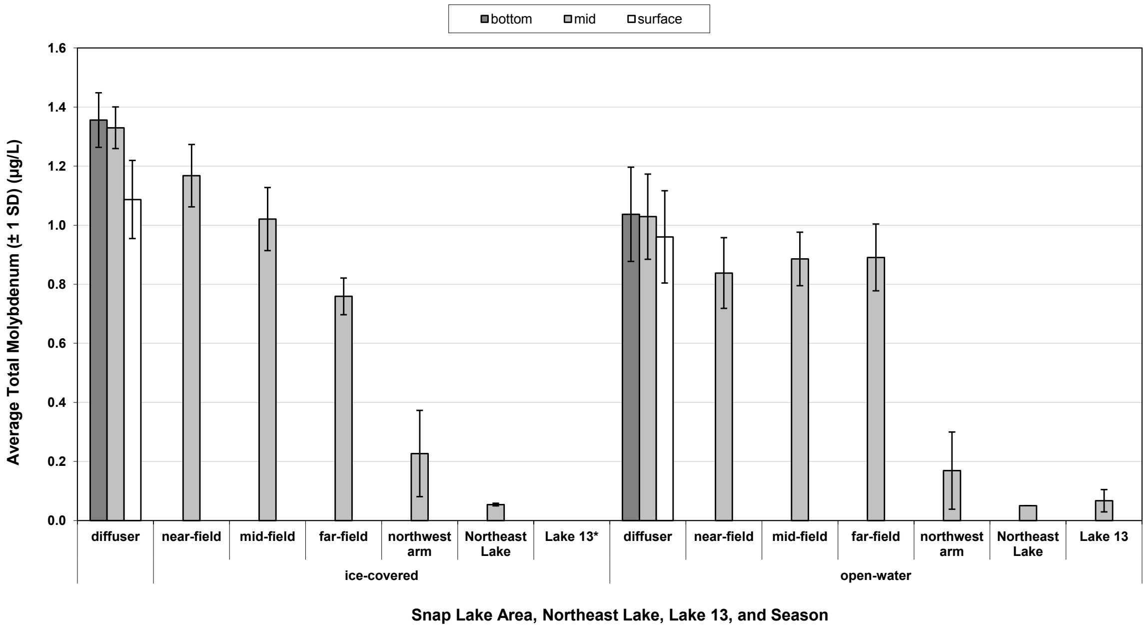
Concentrations of metals were typically similar between Northeast Lake and Lake 13, with the exception of total arsenic and total iron (Figures 3-56 and 3-58). Concentrations of these two metals were higher in Lake 13.

Figure 3-52 Average Total Barium Concentrations in Snap Lake, Northeast Lake, and Lake 13, 2012



* Water samples were not collected from Lake 13 during the ice-covered season.
SD = standard deviation; µg/L = micrograms per litre.

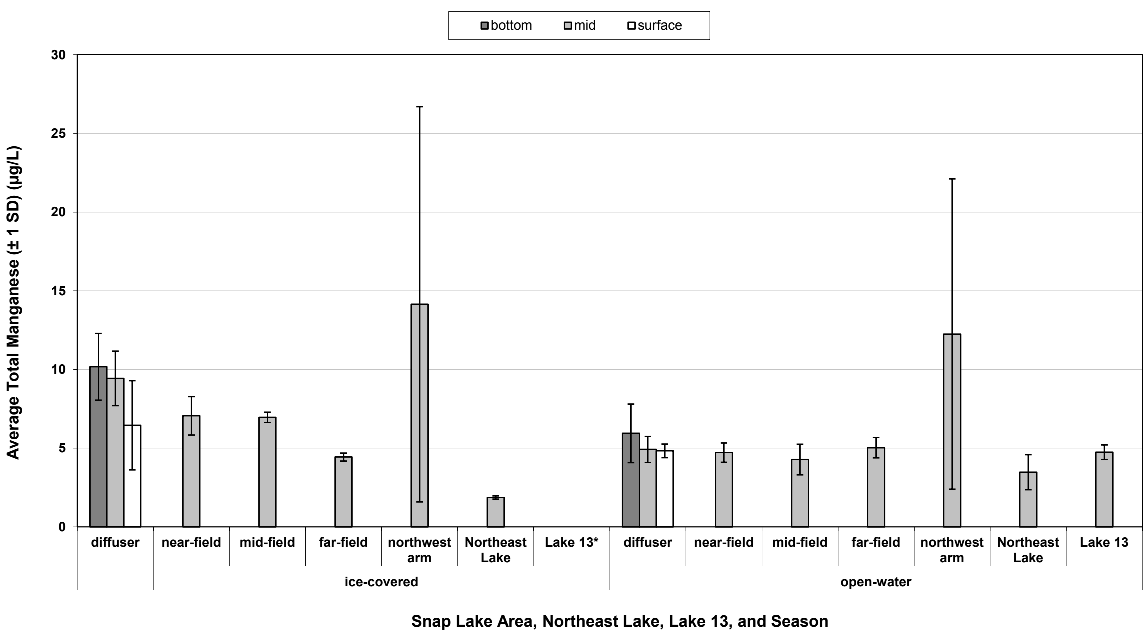
Figure 3-53 Average Total Molybdenum Concentrations (± 1 SD) in Snap Lake, Northeast Lake, and Lake 13, 2012



* Water samples were not collected from Lake 13 during the ice-covered season.

SD = standard deviation; µg/L = micrograms per litre.

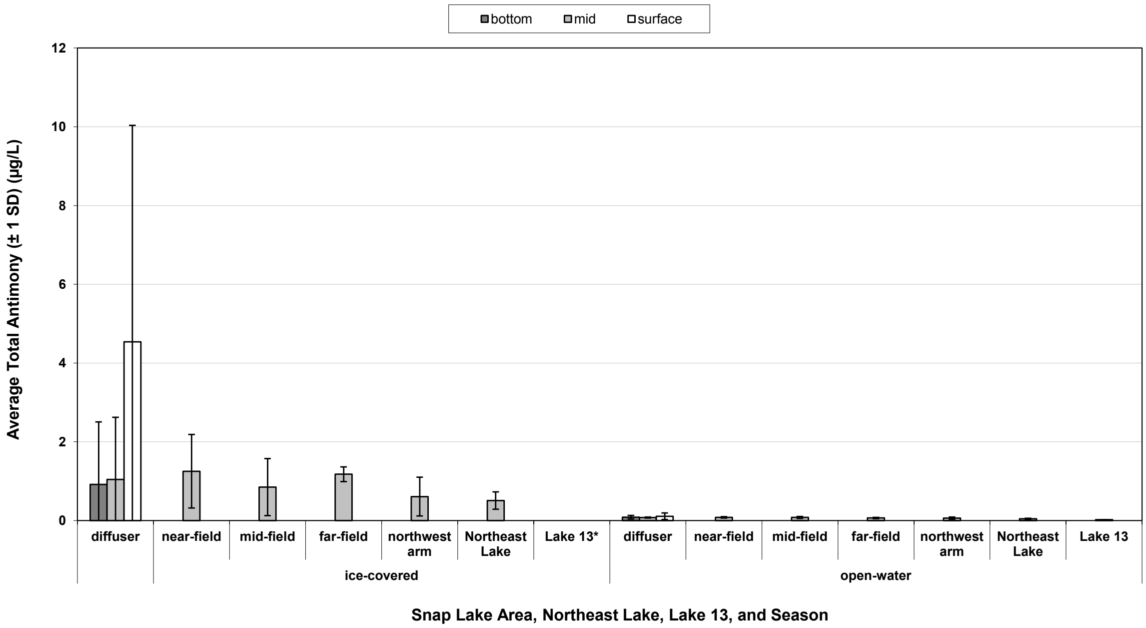
Figure 3-54 Average Total Manganese Concentrations (± 1 SD) in Snap Lake, Northeast Lake, and Lake 13, 2012



* Water samples were not collected from Lake 13 during the ice-covered season.

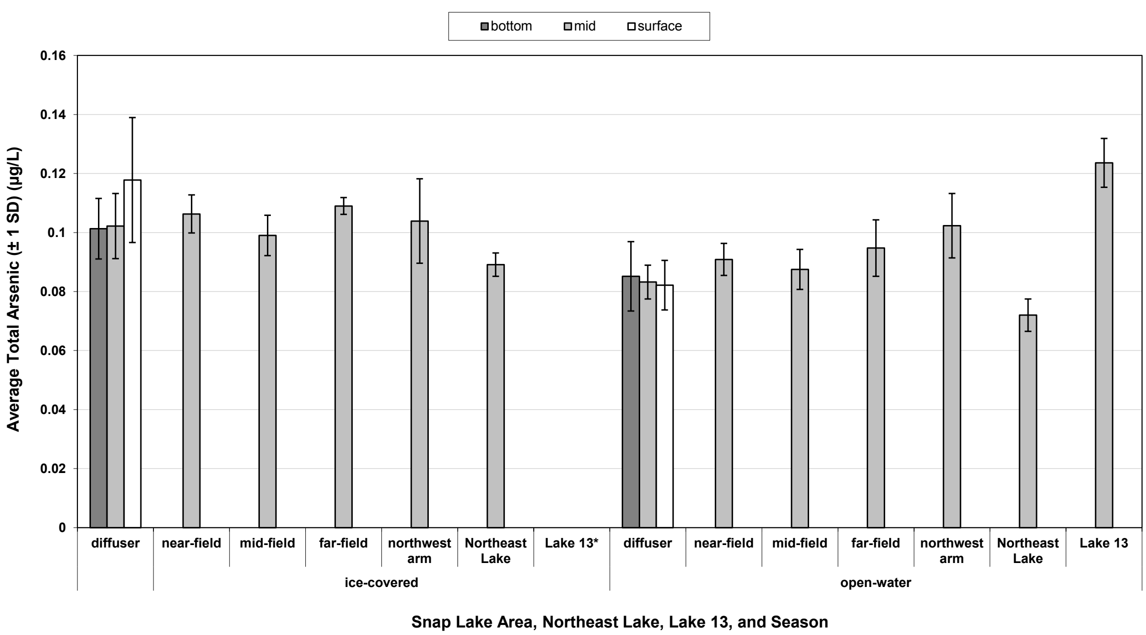
SD = standard deviation; µg/L = micrograms per litre.

Figure 3-55 Average Total Antimony Concentrations (± 1 SD) in Snap Lake, Northeast Lake, and Lake 13, 2012



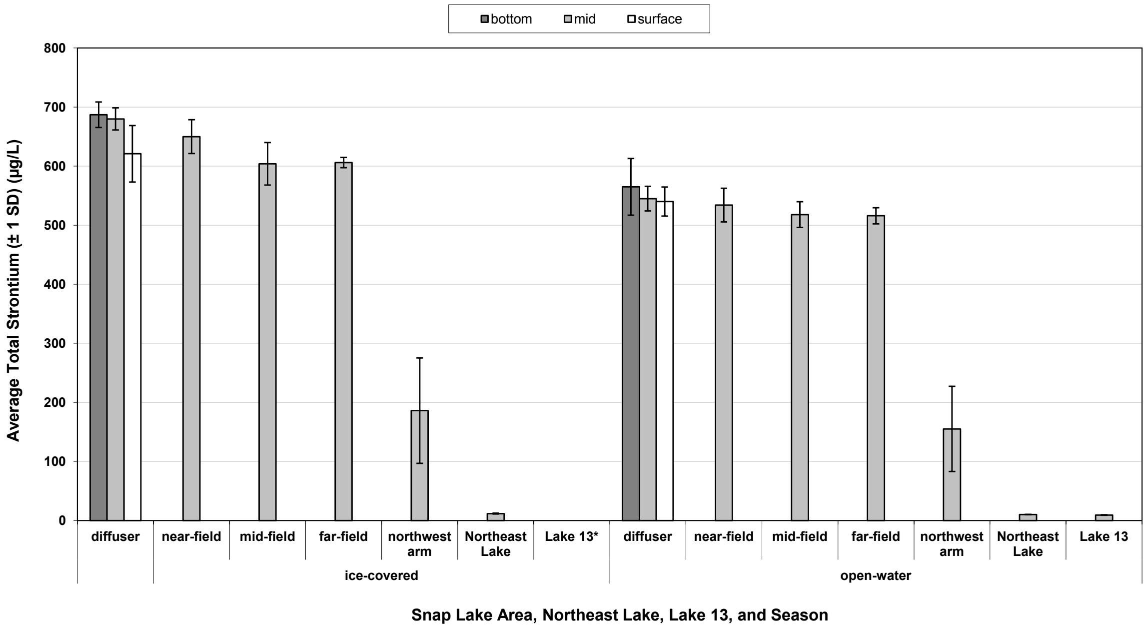
* Water samples were not collected from Lake 13 during the ice-covered season.
SD = standard deviation; µg/L = micrograms per litre.

Figure 3-56 Average Total Arsenic Concentrations (± 1 SD) in Snap Lake, Northeast Lake, and Lake 13, 2012



* Water samples were not collected from Lake 13 during the ice-covered season.
SD = standard deviation; µg/L = micrograms per litre.

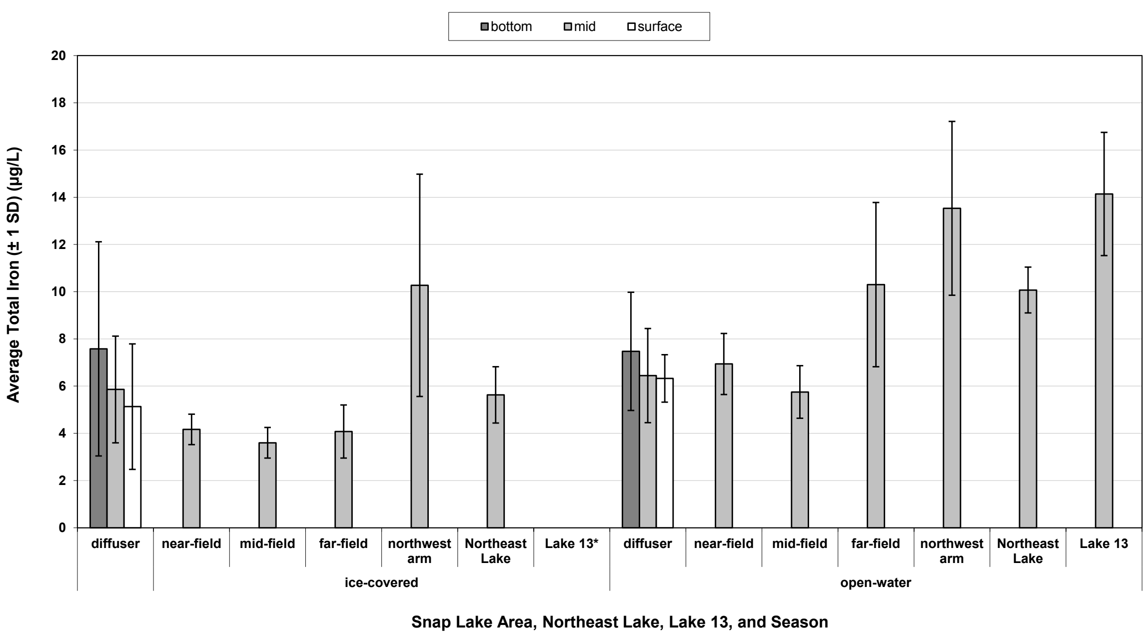
Figure 3-57 Average Total Strontium Concentrations (± 1 SD) in Snap Lake, Northeast Lake, and Lake 13, 2012



* Water sample was not collected from Lake 13 during the ice-covered season.

SD = standard deviation; µg/L = micrograms per litre.

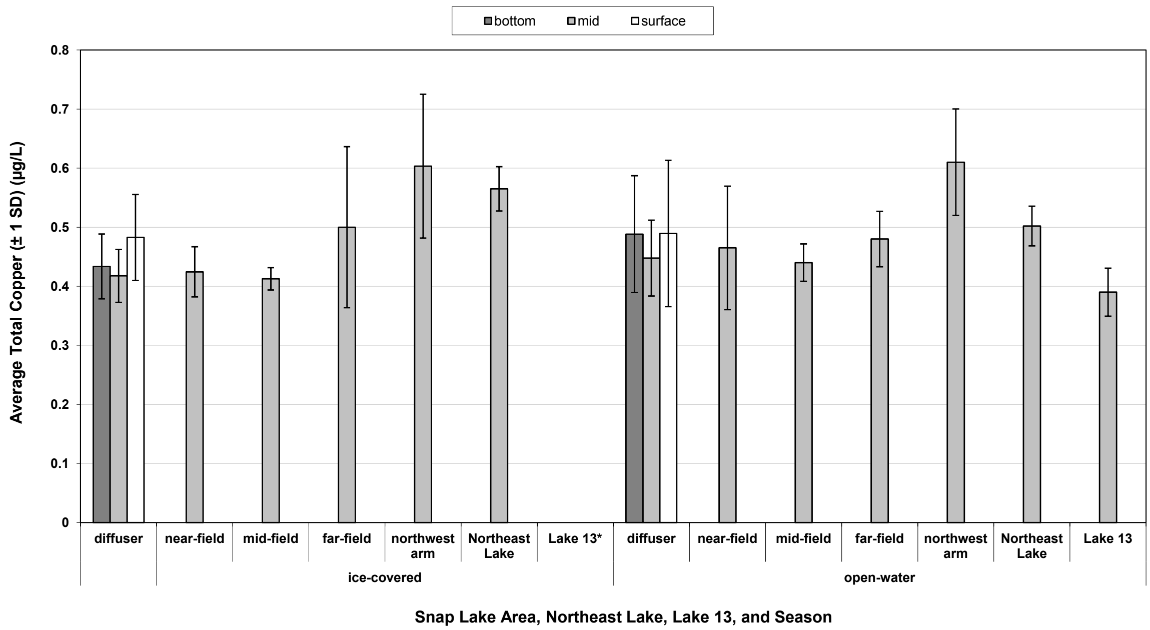
Figure 3-58 Average Total Iron Concentrations (± 1 SD) in Snap Lake, Northeast Lake, and Lake 13, 2012



* Water samples were not collected from Lake 13 during the ice-covered season.

SD = standard deviation; µg/L = micrograms per litre.

Figure 3-59 Average Total Copper Concentrations (± 1 SD) in Snap Lake, Northeast Lake, and Lake 13, 2012



* Water samples were not collected from Lake 13 during the ice-covered season.
SD = standard deviation; µg/L = micrograms per litre.

3.4.5.2 Water Quality Downstream of Snap Lake

King Lake

Water quality at station KING01, which is located 25 km downstream of Snap Lake, was characterized by neutral pH and low alkalinity conditions (Table 3-19). Calculated TDS values ranged from 13 to 18 mg/L in 2012 (Figure 3-60).

Nutrient concentrations were generally near detection limits, and consequently all values were below AEMP benchmarks. Metals concentrations were generally low and below the CCME WQGs and EAR benchmark concentrations (Table 3-19).

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 0.046 (Table 3-20). A *P*-value of less than 0.05 indicates a significant trend in the data. The maximum TDS concentration increased from 12 mg/L in 2005 to 14 mg/L in 2012, which is still considered to be within baseline levels (De Beers 2002). Because station KING01 is located 25 km downstream of Snap Lake, additional volumes of low-TDS waters from the downstream watershed provide substantial dilution to inflows sourced from Snap Lake. Water quality at station KING01 should continue to be monitored. In the EAR, concentrations were conservatively predicted to reach background concentrations within 44 km of Snap Lake during the operating period when maximum concentrations in Snap Lake were predicted to occur.

Table 3-19 Comparison of Water Quality Results at the Downstream Station KING01 and Snap Lake

Parameter	Units	Guideline	Observed Concentrations[b]		
		AEMP Benchmarks (Protection of Aquatic Life)[a]	Type	Snap Lake	Downstream Station (KING01)
Conventional Parameters					
Laboratory pH	unitless	6.5 - 9.0	range	6.8 to 7.7	7.0 to 7.1
Total dissolved solids, calculated (Standard Methods)	mg/L	-	max	279	14
Ions					
Chloride	mg/L	120	max	**121**	3
Fluoride	mg/L	0.12	max	**0.18**	0.06
Sodium	mg/L	-	max	31	1
Calcium	mg/L	-	max	62	2
Magnesium	mg/L	-	max	7.5	0.9
Sulphate	mg/L	-	max	24	2
Alkalinity, as $CaCO_3$	mg/L	-	range	6 to 32	6 to 7
Hardness, as $CaCO_3$	mg/L	-	range	26 to 186	7 to 10
Nutrients and Carbons					
Nitrate, as N	mg-N/L	2.93	max	**3.22**	0.01
Nitrite, as N	mg-N/L	0.06	max	0.029	<0.002
Ammonia, as N	mg-N/L	0.74 to 17.5[c]	max	0.32	<0.005
Total phosphorus	mg-P/L	-	max	0.018	0.003
Total Metals					
Aluminum	µg/L	100[d]	max	15	12
Arsenic	µg/L	5	max	0.2	0.1
Barium	µg/L	-	max	29	4
Boron	µg/L	1,500	max	53	4
Cadmium	µg/L	0.36	max	0.07	0.04
Chromium	µg/L	8.9	max	0.3	0.1
Hexavalent chromium	µg/L	2.1	max	1.2	<1
Copper	µg/L	7.9	max	0.8	0.7
Iron	µg/L	300	max	19	19
Lead	µg/L	1 [e]	max	0.07	0.01
Lithium	µg/L	-	max	11	1
Manganese	µg/L	-	max	39	7
Mercury (Flett)	µg/L	0.026	max	0.0012	<0.0005
Molybdenum	µg/L	73	max	1.54	<0.05
Nickel	µg/L	25[e]	max	2.17	0.47
Selenium	µg/L	1	max	0.044	<0.04
Silver	µg/L	0.1	max	<0.005	<0.005
Thallium	µg/L	0.8	max	<0.01	<0.01
Uranium	µg/L	-	max	0.2	0.02
Zinc	µg/L	30	max	9	2

(a) AEMP benchmarks are: WQGs from the Canadian Council of Ministers of the Environment (CCME) (1999 with updates to 2012) 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).

(b) Observed concentrations within the 2012 reporting period (January 1, 2012 to September 30, 2012). **Bold** values are above the relevant benchmarks.

(c) Ammonia WQG is pH and water temperature dependent. Range of the WQG shown is based on a range of laboratory pH from 6.9 to 7.1 and a range of water temperature from 0.4 to 16.8 °C, which were observed in Snap Lake during the 2012 reporting periods. The WQG was calculated based on an individual pH and water temperature for each sample with the final value expressed as ammonia as nitrogen. When water temperature was not available in an individual sample, the average of historical water temperature for the month in the region was used to calculate the ammonia WQG.

(d) Aluminum WQG is pH dependent. The WQG shown here is based on a range of pH from 6.9 to 7.1, which was observed in Snap Lake during the 2012 reporting period. The WQG was calculated based on the individual pH for each sample.

(e) Lead and nickel WQGs are hardness dependent. The range of the WQGs shown here was based on a range of hardness from 7.4 to 10 mg/L, which was observed in Snap Lake during the 2012 reporting period. The WQG was calculated based on the individual hardness for each sample.

Flett = Flett Research Limited; - = not applicable; N = nitrogen; calc'd = calculated; $CaCO_3$ = calcium carbonate; max = maximum; <= less than; ≤ = less than or equal to; µg/L= microgram per litre; mg/L= milligram per litre; °C = degrees Celsius.

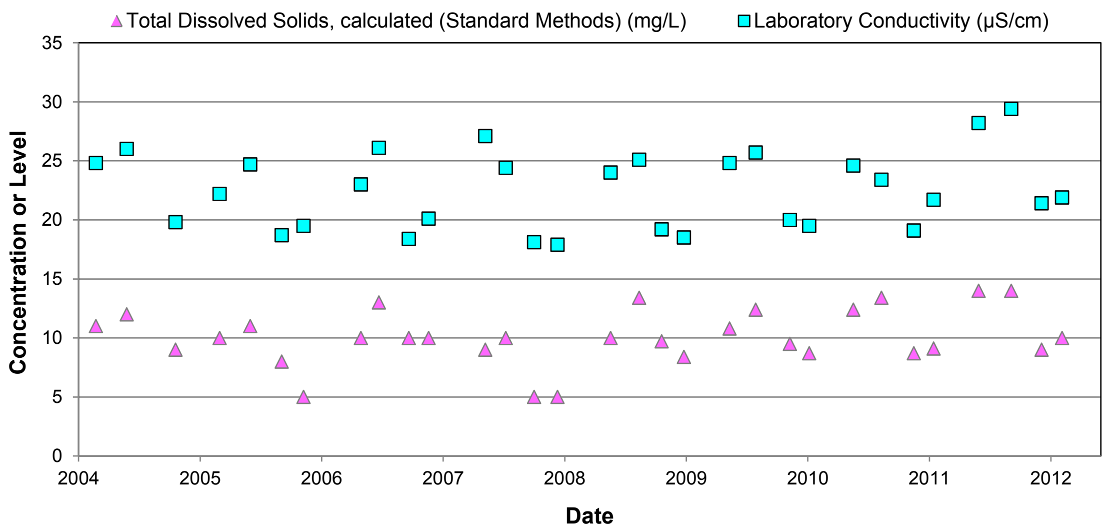
Table 3-20 Summary of Temporal Trend for Total Dissolved Solids at Station KING01 Using the Seasonal Kendall Test

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	32	1.996	0.046	↑

Note: The Seasonal Kendall Test was run using SYSTAT 13.1.00.5 (SYSTAT 2009); ↑ = an increasing trend;% = percent; n = sample count.

(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.

Figure 3-60 Concentrations of Total Dissolved Solids and Conductivity at the Downstream Station KING01, 2005 to 2012



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

Summary of the Downstream Lakes Special Study

A Downstream Lakes Special Study was conducted as part of the 2012 AEMP to document the spatial extent of treated effluent downstream of Snap Lake. Field water column profiles and water samples were collected along transects to assess the lateral and vertical extent of treated effluent in downstream lakes, including two lakes immediately downstream of Snap Lake (i.e., DSL1 and DSL2) and Lac Capot Blanc (i.e., LCB).

Concentrations of TDS and other Mine-related parameters (e.g., conductivity, major ions, nitrate), decreased with distance downstream, consistent with EAR predictions (Section 12.2). In 2012, evidence of the treated effluent was measured throughout DSL1 and DSL2, and near the inlet of Lac Capot Blanc. Field conductivity decreased from 240 µS/cm at the inlet of Lac Capot Blanc, to near background levels at the two outlets of Lac Capot Blanc, which are located 3.3 and 4.7 km northeast of the inlet (Section 12.2).

Treated effluent extended approximately 650 m from the inlet of Lac Capot Blanc and approximately 5.8 km downstream from Snap Lake's outlet in 2012 (Section 12.2). Based on the 2012 conductivity values, the area influenced by treated effluent has increased since 2011, when it decreased to background levels within 50 m of the inlet of Lac Capot Blanc.

3.4.5.3 Summary of Key Question 4

Spatial and seasonal patterns were observed for some water quality parameters in Snap Lake. Horizontal patterns included gradual declines in concentration with increasing distance away from the diffuser for TDS and water quality parameters related to Mine activity (conductivity, most major ions, nitrogen-nutrients [nitrate, nitrite, ammonia], and eight metals [barium, boron, lithium, molybdenum, nickel, rubidium, strontium, and uranium]). However, concentration gradients within the main basin of Snap Lake for these parameters were less prominent in 2012 compared to gradients observed in the first four years of minewater discharges to Snap Lake (i.e., 2004 and 2007). The increase in concentrations has now become widespread throughout the main body of Snap Lake.

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 are now higher than those observed at Northeast Lake, and the increased concentration was evident close to the northwest arm's narrow connection to the main basin again in 2012.

Seasonal differences between ice-covered and open-water conditions were less prominent in 2012 compared to those observed 2004 to 2007. The reduction in the range of seasonal and spatial differences is attributed to the greater exposure of treated effluent discharge within Snap Lake.

Concentrations of TDS were at background levels near King Lake, which is 25 km downstream. Results from the Downstream Lakes Special Study (Section 12.2) showed evidence of an influence of the treated effluent throughout lakes DSL1 and DSL2, and near the inlet of Lac Capot Blanc in 2012. Concentrations of Mine-related constituents reached background concentrations approximately 6 km downstream of Snap 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 to provide additional information for assessing variability of water quality within lakes as well as to be used in refining the environmental assessment. 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 2012 for three inland lakes, IL3, IL4, and IL5, as well as the two major tributaries to Snap Lake, Streams S1 and S27, to identify sensitivity to acidification and any changes resulting from potentially acidifying deposition from Mine emissions. The acidification assessment will be re-visited in 2013 using updated air modelling information and water quality data to determine whether the results from the 2009 assessment remain valid.

Acid sensitivity for each waterbody was evaluated by comparing mean alkalinity concentrations to a scale presented by Saffran and Trew (1996, Table 3-21), while concentrations of sulphate, nitrate, pH, alkalinity and base cations were examined for trends which might be indicative of acidification as a result of Mine emissions.

Table 3-21 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.

3.4.6.1 Inland Lakes

The three inland lakes monitored in 2012 were characterized by low TDS, neutral to slightly acidic pH, and low alkalinity (Table 3-22), consistent with the water quality observed in 2011 (De Beers 2012a). Dissolved oxygen concentrations measured in the inland lakes were within WQGs for the protection of aquatic life. One field pH value in Lake IL4 was just below the WQG range. Concentrations of major ions and nitrogen parameters in the inland lakes were well below the CCME WQGs (Table 3-22). The 2012 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 10 mg/L as $CaCO_3$).

Table 3-22 Summary of Selected Water Quality Results for Three Inland Lakes

Parameter Name	Unit	AEMP Benchmark (Protection of Aquatic Life) [a]	Observed Concentrations[b]			
			Type	IL3	IL4	IL5
Field Parameter						
pH	unitless	-	min, mean	**6.4**, 6.5	**6.4, 6.4**	6.5, 6.6
Dissolved oxygen	mg/L	6.5, 9.5 [c]	min, mean	8.7, 9.9	9.3, 11.1	9.2, 10.6
Conductivity	μS/cm	-	mean	33.5	22.5	32.5
Conventional Parameters						
Laboratory pH	unitless	6.5 to 9.0	mean	6.8	6.6	7.1
Total dissolved solids, calculated (Standard Methods)	mg/L	-	mean	8.6	7.4	21.9
Conductivity	μS/cm	-	mean	19.3	20.4	44.8
Major Ions						
Chloride	mg/L	120	mean	0.3	0.5	1.7
Fluoride	mg/L	0.12	mean	0.06	0.04	0.1
Hardness, as $CaCO_3$	mg/L	-	mean	7.5	8.1	15.8
Sodium	mg/L	-	mean	0.5	0.9	1.4
Sulphate	mg/L	-	mean	1.4	0.3	6.14
Total alkalinity, as $CaCO_3$	mg/L	-	mean	4	5	9.7
Nutrients						
Nitrate, as N (calculated)	mg-N/L	2.93	mean	0.022	0.023	0.017
Nitrite, as N	mg-N/L	0.06	mean	0.003	0.005	0.001
Total Ammonia, as N	mg-N/L	2.45 to 38.1[d]	mean	0.014	0.053	0.008

(a) Water Quality Guidelines (WQGs) are from the Canadian Council of Ministers of the Environment (CCME) (1999).
(b) Observed concentrations within the 2012 reporting period (January 1, 2012 to September 30, 2012). Bold values are above the relevant WQGs or outside of the guideline range for pH.
(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 laboratory pH from 6.3 to 7.3 and a range of water temperature from 1.0 to 16.9 °C, which were observed in Snap Lake during the 2012 reporting period. The WQG was calculated based on an individual pH and water temperature for each sample with the final value expressed as ammonia as nitrogen.
$CaCO_3$ = calcium carbonate; N = nitrogen; - = not applicable; mg/L = milligram per litre; mg-N/L = milligram as nitrogen per litre; μS/cm = microSiemens per centimetre.

Comparison to Key Acidification Indicator Parameters

Sulphate concentrations in IL3 and IL4 in 2012 were lower than baseline concentrations (i.e., concentrations measured in 1999 and 2002) (Figure 3-61). Overall, concentrations of sulphate in IL3 and IL4 have remained relatively stable since annual monitoring of the inland lakes began five years ago; however, concentrations appear to have increased slightly in IL4 since 2010, while decreasing slightly in IL3 during the same time period. 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.

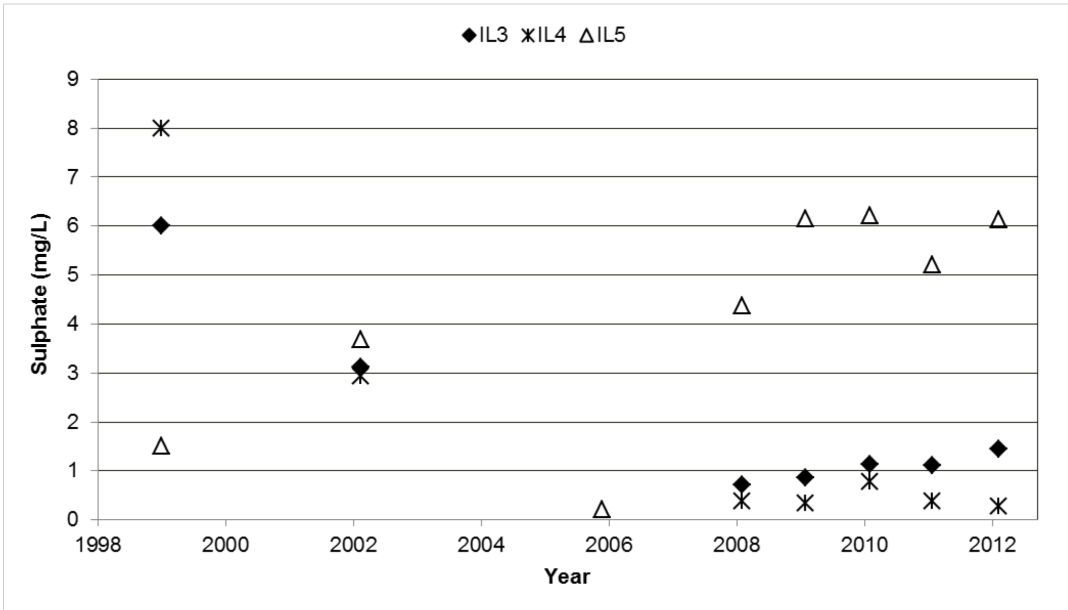
Nitrate concentrations in the inland lakes in 2012 were either at or near baseline concentrations (Figure 3-62). Concentrations of nitrate were elevated in IL5 between 2006 and 2009, but were much lower from 2010 to 2012.

Base cation concentrations were consistently higher than baseline concentrations in the inland lakes over the 13-year-sampling record (Figure 3-63), and have remained relatively stable since annual monitoring of the inland lakes began five years ago. 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 pH and total alkalinity in 2012 were comparable to those observed between 2008 and 2011 in the inland lakes (Figures 3-64 and 3-65). Compared to baseline conditions, pH measurements were higher in all three lakes, particularly IL5, and have remained relatively consistent in each lake since annual monitoring began in 2008.

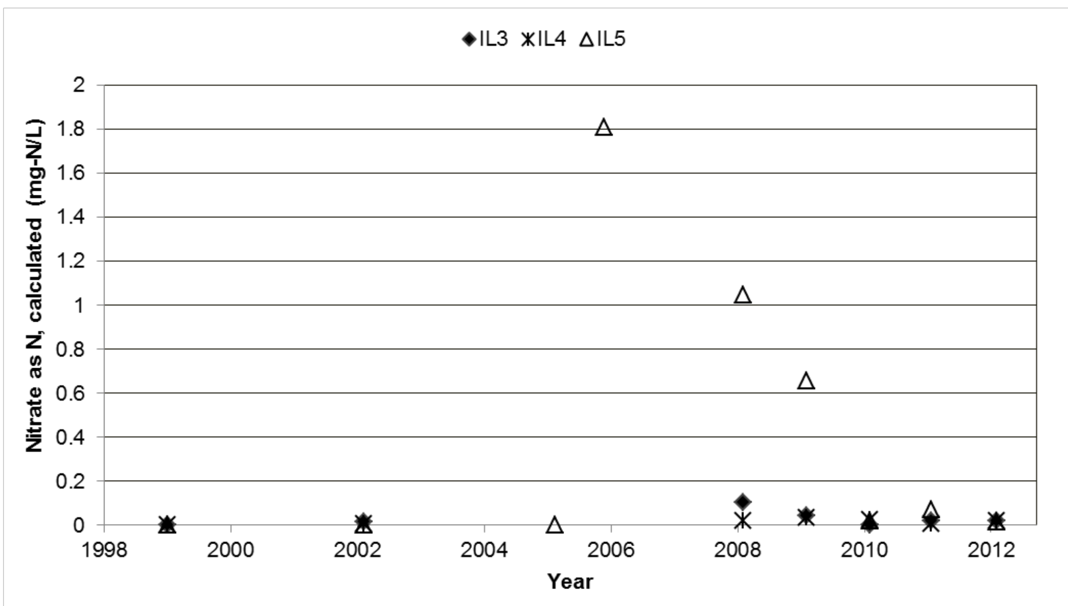
Based on the 2012 data, there was no strong evidence of acidification of these lakes. These results are consistent with previous assessments (De Beers 2010a, 2011a, 2012a), which concluded that there is limited potential for acidification of these lakes due to emissions from the Mine. However, data shown in Figures 3-61 to 3-65 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, pH, and alkalinity are elevated compared to baseline. Within-lake fluctuations of sulphate, nitrate, pH, base cations, and alkalinity observed in IL3 and IL4, appear to be within the normal range of variation seen since monitoring began five years ago but are generally different from baseline conditions.

Figure 3-61 Sulphate Concentrations in Inland Lakes 3, 4, and 5, 1999 to 2012



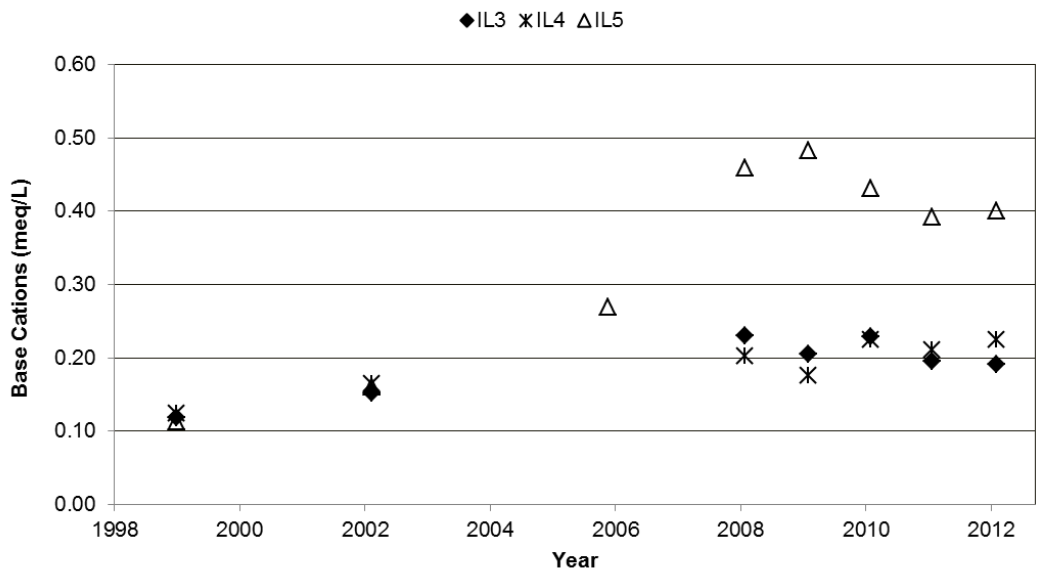
Note: Mean concentrations are shown for 2008 to 2012; values prior to 2008 represent one discrete sample.
mg/L = milligram per litre

Figure 3-62 Nitrate Concentrations in Inland Lakes 3, 4, and 5, 1999 to 2012



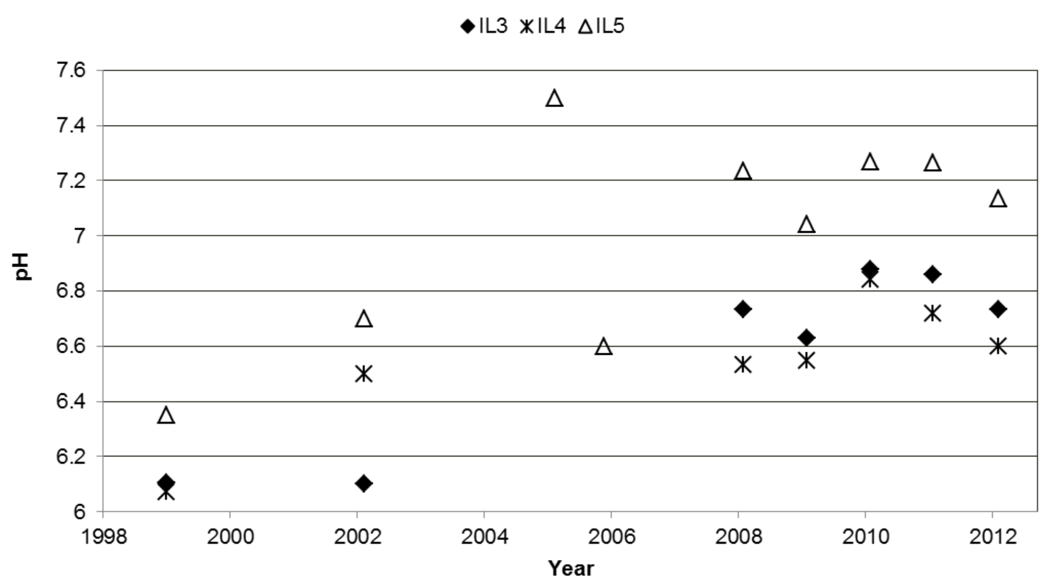
Note: Nitrate+ nitrite concentrations were plotted for the 1999 data, because nitrate values were not available.
Mean concentrations are shown for 2008 to 2012; values prior to 2008 represent one discrete sample.
N = nitrogen; mg-N/L= milligram as nitrogen per litre.

Figure 3-63 Sum of Base Cations in Inland Lakes 3, 4, and 5, 1999 to 2012



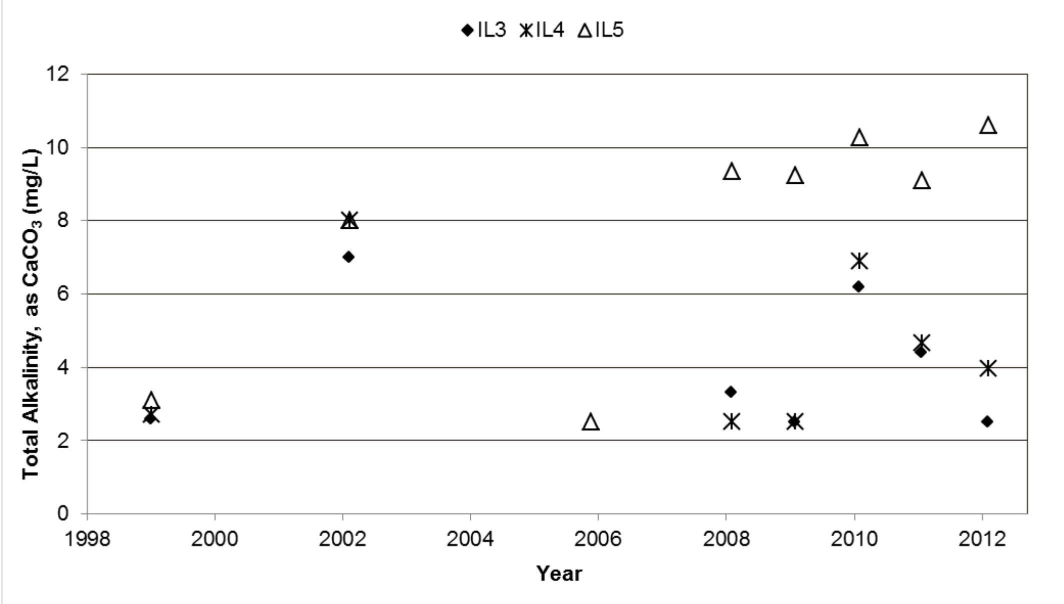
Note: Mean concentrations are shown for 2008 to 2012; values prior to 2008 represent one discrete sample.
meq/L = milliequivalent per litre.

Figure 3-64 Concentrations of Laboratory pH in Inland Lakes 3, 4, and 5, 1999 to 2012



Note: Mean concentrations are shown for 2008 to 2012; values prior to 2008 represent one discrete sample.

Figure 3-65 Total Alkalinity in Inland Lakes 3, 4, and 5, 1999 to 2012



Note: Mean concentrations are shown for 2008 to 2012; values prior to 2008 represent one discrete sample.
$CaCO_3$ = calcium carbonate; mg/L = milligrams per litre.

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, and 2011). 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 alkalinity, low concentrations of major ions, and slightly acidic waters. Aluminum and iron have typically been higher in S1 (De Beers 2002, 2006, 2007a, 2008a, 2009, 2010, 2011a, 2012b) and S27 (De Beers 2002) compared to Snap Lake. In 2012, the maximum concentrations of all water quality parameters measured from Streams S1 and S27 were below AEMP benchmarks with some exceptions (Table 3-23). The maximum iron concentration was above the WQG in S1 (Table 3-23). Iron concentrations have been consistently high in Stream S1 since 1999. The maximum aluminum concentration was above the WQG in Stream S27 (Table 3-23). Aluminum was also above the WQG in the baseline (i.e., 1999) metals data collected from S27 (De Beers 2002).

The ranges in pH, total alkalinity, and sulphate values in Streams S1 and S27 in 2012 were similar to baseline values (Figures 3-66 to 3-69) 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 2012.

Table 3-23 Summary of Selected Water Quality Results for Streams S1 and S27, 2012

Parameter	Units	AEMP Benchmarks (Protection of Aquatic Life)[a]	Observed Concentrations[b]		
			Type	Stream S1	Stream S27
Field Parameters					
Dissolved oxygen	mg/L	6.5, 9.5[c]	min	**7.7**	**5.7**
Conventional Parameters					
Laboratory pH	unitless	6.5 to 9.0	range	6.1 to 7.2	6.4 to 7.0
Total dissolved solids, calculated (Standard Methods)	mg/L	-	max	21	17
Major Ions					
Calcium	mg/L	-	max	3	2
Chloride	mg/L	120	max	3	2
Fluoride	mg/L	0.12	max	0.06	<0.05
Hardness, as $CaCO_3$	mg/L	-	range	6 to 12	6 to 9
Magnesium	mg/L	-	max	1.2	1.1
Sodium	mg/L	-	max	2	1
Sulphate	mg/L	-	max	2	2
Total alkalinity, as $CaCO_3$	mg/L	-	range	<5 to 12	<5 to 7
Nutrients					
Nitrate, as N	mg-N/L	2.93	max	0.71	0.91
Nitrite, as N	mg-N/L	0.06	max	0.004	0.004
Ammonia, as N	mg-N/L	2.79 to 139[d]	max	0.07	0.14
Total Metals					
Aluminum	µg/L	5 to 100[e]	max	83	**105**
Arsenic	µg/L	5	max	0.1	0.1
Boron	µg/L	1,500	max	10	4
Cadmium	µg/L	0.36	max	0.01	0.02
Chromium	µg/L	8.9	max	0.34	0.18
Copper	µg/L	7.9	max	1.3	2.6
Iron	µg/L	300	max	**725**	282
Lead	µg/L	1[f]	max	0.04	0.02
Mercury[g]	µg/L	0.026	max	0.002	<0.02
Molybdenum	µg/L	73	max	0.35	0.09

Table 3-23 Summary of Selected Water Quality Results for Streams S1 and S27, 2012

Parameter	Units	AEMP Benchmarks (Protection of Aquatic Life)[a]	Observed Concentrations[b]		
			Type	Stream S1	Stream S27
Nickel	µg/L	25[f]	max	0.8	0.6
Selenium	µg/L	1	max	<0.04	<0.04
Silver	µg/L	0.1	max	<0.005	<0.005
Thallium	µg/L	0.8	max	<0.01	<0.01
Uranium	µg/L	15	max	0.04	0.05
Zinc	µg/L	30	max	7	2

(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 within the 2012 reporting period (January 1, 2012 to September 30, 2012). Bold values are above the relevant benchmarks.

(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 laboratory pH from 6.1 to 7.2 and a range of water temperature from 0.1 to 20.1 °C, which were observed in Streams S1 and S27 during the 2012 reporting periods. The WQG was calculated based on an individual pH and water temperature for each sample with the final value expressed as ammonia as nitrogen. When water temperature was not available in an individual sample, the average of historical water temperatures for the month in the region was used to calculate the ammonia WQG.

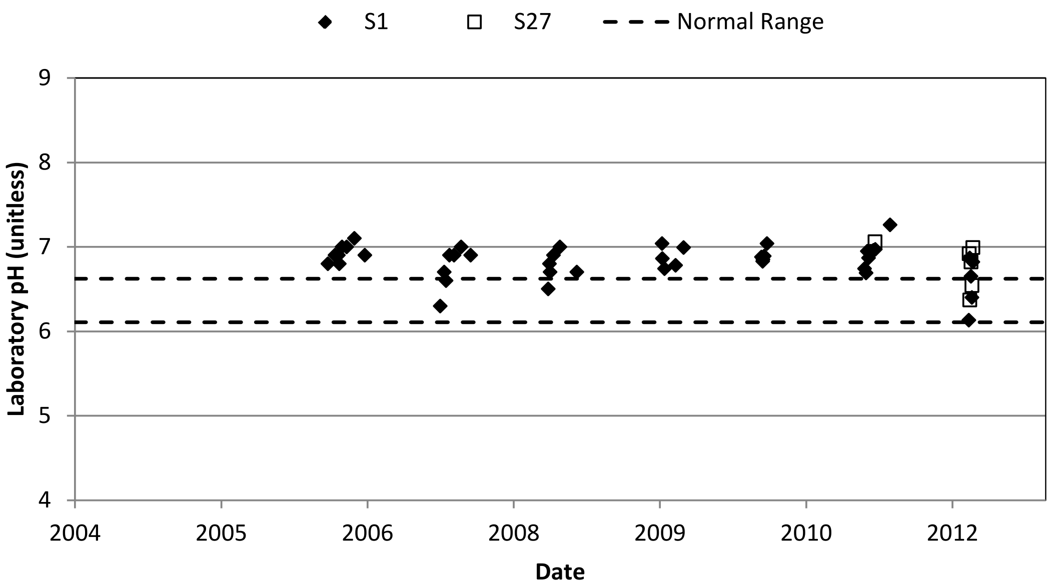
(e)Aluminum WQG is pH dependent. The WQG shown here is based on a range of pH from 6.1 to 7.2, which was observed in Streams 1 and 27 during the 2012 reporting periods. The WQG was calculated based on the individual pH for each sample.

(f) Lead and nickel WQGs are hardness dependent. The range of the WQGs shown here was based on a range of hardness from 25.6 to 186 mg/L, which was observed in Streams 1 and 27 during the 2012 reporting period. The WQG was calculated based on the individual hardness for each sample.

(g) Mercury results analyzed by Flett Research Ltd. were included for Snap Lake and Stream 1. For Stream 27, the results analyzed by ALS were included.

N = nitrogen; $CaCO_3$ = calcium carbonate; N = nitrogen; - = not applicable; <= less than; ≤ = less than or equal to; max = maximum; min = minimum; µg/L = microgram per litre; mg/L = milligram per litre; mg-N/L = milligram as nitrogen per litre; mg-P/L = milligram as phosphorus per litre;% = percent.

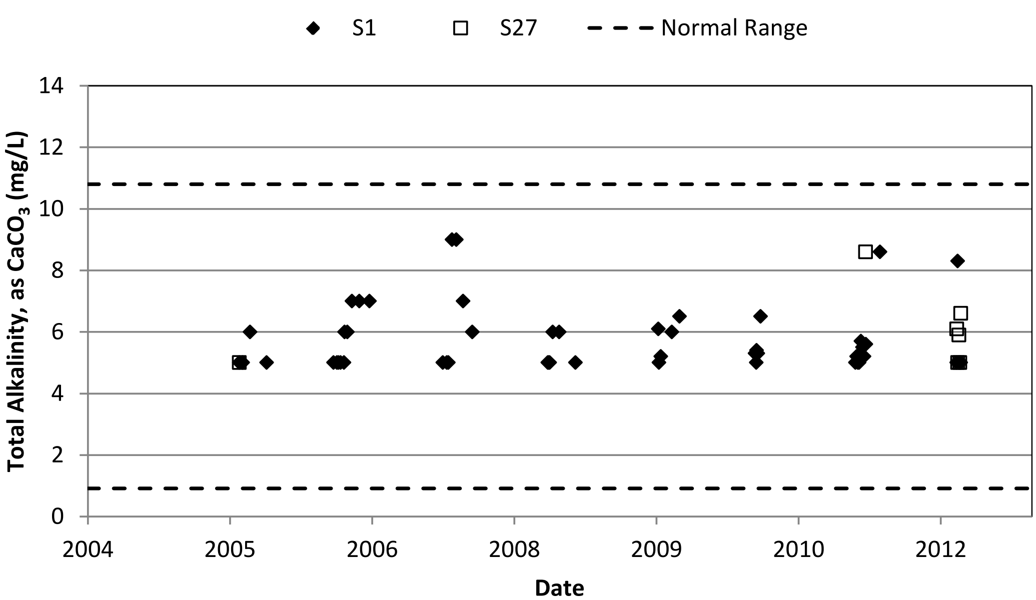
Figure 3-66 Concentrations of Laboratory pH in Stream 1 and Stream 27, 2004 to 2012



Note: Normal range based on data collected from 1999 to 2002, with the upper and lower range calculated as the mean ± 2 standard deviations.

S1 = Stream S1; S27 = Stream S27.

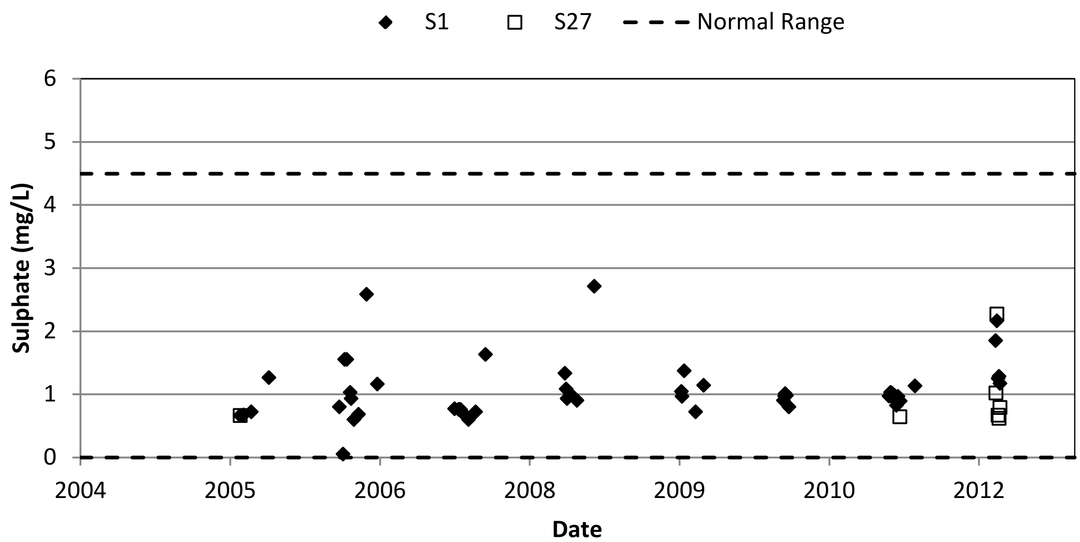
Figure 3-67 Concentrations of Total Alkalinity in Stream 1 and Stream 27, 2004 to 2012



Note: Normal range based on data collected from 1999 to 2002, with the upper and lower range calculated as the mean ± 2 standard deviations.

S1 = Stream S1; S27 = Stream S27; $CaCO_3$ = calcium carbonate; mg/L = milligrams per litre.

Figure 3-68 Concentrations of Total Sulphate in Stream 1 and Stream 27, 2004 to 2012



Note: Normal range based on data collected from 1999 to 2002, with the upper and lower range calculated as the mean ± 2 standard deviations.

S1 = Stream S1; S27 = Stream S27; mg/L = milligrams per litre.

3.4.6.3 Summary of Key Question 5

Water quality data collected from Snap Lake and the water intake station (SNP 02-15) were compared to Canadian drinking WQGs for aesthetic objectives (AO) and maximum acceptable concentrations (MAC) (Table 3-24). Drinking WQGs for human consumption are more stringent than wildlife WQGs (livestock watering guidelines: CCME 1999). Therefore, water considered safe for humans to drink would also be considered safe for wildlife consumption. If results were above the MACs, an attempt was made to determine the relevance of the elevated results to potential for risk to human health. Where appropriate, this analysis involved additional comparison to average conditions and baseline conditions in Snap Lake, consideration of the frequency, duration, and location of the elevated result, treatment practices, and the potential source of that parameter.

3.4.7 Key Question 6: Is water from Snap Lake safe to drink?

Water quality data collected from Snap Lake and the water intake station (SNP 02-15) were compared to Canadian drinking WQGs for Aesthetic Objectives (AO) and Maximum Acceptable Concentrations (MAC) (Table 3-24). 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 the MACs, an attempt was made to determine the relevance of the elevated results to potential for risk to human health. Where appropriate, this analysis involved additional comparison to average conditions and baseline conditions in Snap Lake, consideration of the frequency, duration and location of the elevated result, treatment practices, and the potential source of that parameter.

Concentrations of most water quality parameters collected from Snap Lake and SNP 02-15 were below MACs (Health Canada 2012), with the exception of *E. coli*, total coliforms, and total antimony (Table 3-24). These parameters are discussed in more detail below.

Escherichia coli and Total Coliforms

In Snap Lake, the maximum *E. coli* value of 1 most probable number per 100 millilitre (MPN/100 mL) was above the MAC of no detectable *E. coli* per 100 mL. The maximum was measured in the two samples collected at surface and mid-depth of diffuser station, SNP 02-20f, on August 12, 2012. However, *E. coli* were not detected in the remaining water samples collected during the 2012 AEMP (Appendix 3C; Table 3C-1). In the treated effluent, the maximum *E. coli* value was 8 MPN/100 mL (Appendix 3D; Table 3D-2).

A maximum total of 10 colony forming units per 100 millilitres (cfu/100 mL), which was above the MAC of no detectable coliforms per 100 mL, was measured in one sample collected at SNP 02-15 on October 1, 2012 (Appendix 3H; Table 3H-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 suggest that water from Snap Lake should be disinfected before human consumption, which is 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 (micrometer) 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 is then 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.

Total Antimony

Concentrations of ions, nutrients, metals and organic parameters from Snap Lake and SNP 02-15 were below drinking WQGs, with the exception of total antimony. Approximately 4% of the 2012 water samples from Snap Lake contained antimony at concentrations above the MAC of 6 μg/L. The antimony concentrations that were above the MAC were measured near the surface at the diffuser stations during the late ice-covered conditions (i.e., April and May). The maximum concentration of 16.8 μg/L was measured at diffuser station SNP 02-20d on May 13, 2012 (Appendix 3C; Table 3C-1). Average antimony concentrations in each area of Snap Lake (i.e., diffuser, near-field, mid-field, far-field, and northwest arm) were below the MAC in 2012. The range in average total antimony concentrations was 0.06 μg/L (northwest arm) to 4.5 μg/L (diffuser) (Table 3-24). For dissolved antimony, concentrations ranged from 0.04 μg/L to 1.1 μg/L.

Antimony may enter the aquatic environment by way of natural weathering of rocks, runoff from soils, or in mining effluents (Health Canada 1997). As well, leaching from piping can be a source of antimony (Health Canada 1997). Elevated antimony concentrations have been historically measured near the diffusers during ice-covered conditions between 2007 and 2011 (Appendix 3F; Figure 3F-25). However, antimony concentrations were not correlated with conductivity (Appendix 3F), and temporal trends were not identified (Table 3-25). Antimony concentrations in the treated effluent ranged from 0.12 to 0.38 μg/L in 2012, well below concentrations at the diffuser and the MAC for antimony. Thus, treated effluent was likely not the source of the observed concentrations above the MAC. Other possible sources include leaching of antimony from the diffuser structure, physical (e.g., re-suspension) or chemical (e.g., oxidation-reduction) processes that could have influenced antimony concentrations in the water column and sediment (Section 4.0).

Results from the QA/QC assessment indicated that antimony concentrations in 2012 may have been affected by contamination or analytical interference, thus the 2012 antimony data should be interpreted with this limitation in mind (Appendix 3A). Concentrations of antimony were detected in all of the equipment blank samples collected during the 2012 AEMP program (Appendix 3A). In one blank sample, the total antimony concentration was 14 μg/L. Although that sample was not collected at the time the maximum total antimony was observed, it does indicate that contamination or analytical interference were present. As well, there were notable differences (i.e., relative percent difference (RPD) greater than 20%), for total antimony in both the duplicate and split samples collected. Finally, many of the dissolved antimony concentrations were greater than the total antimony concentrations in the blank samples. Due to these issues, which make the antimony data uncertain, one of the recommendations from the QA/QC assessment was to continue to investigate antimony contamination and analysis methods in 2013.

Based on the measured antimony concentrations in the northwest arm and intake waters, the camp workers who drink the water are not at risk, as concentrations were well below the MAC. The maximum concentration in the northwest arm was 0.06 μg/L (i.e., two orders of magnitude below the MAC); the dissolved concentrations would be even lower after the filtration process in the drinking water treatment was applied. Raw and potable water at the Mine have been tested and reported to the local Health Authority (De Beers 2013b). Total antimony concentrations have been well below the MAC in those samples. Therefore, the overall drinkability of Snap Lake water for the Mine workers was not affected.

Antimony should continue to be investigated through follow-up QA/QC measures. If the QA/QC investigation indicates that the observed values above the MAC are real, total antimony should be added to the modelling parameter suite to investigate whether any physical or chemical processes are influencing antimony concentrations and what the maximum concentrations throughout the lake are predicted to be throughout Mine operations.

Table 3-24 Comparison of Snap Lake Water Quality to Canadian Drinking Water Quality Guidelines

Parameter	Units	Canadian Drinking Water[(a)]	Type	Observed Data[(b)]	
				Snap Lake	SNP 02-15
Conventional Parameters					
Laboratory-pH	unitless	6.5 to 8.5	range	6.8 to 7.7	7.0 to 7.4
Total dissolved solids, calculated (Standard Methods)	mg/L	≤500 (AO)	max	279	171
Major Ions					
Chloride	mg/L	≤250 (AO)	max	121	75
Cyanide	mg/L	200	max	-	<0.002
Fluoride	mg/L	1.5	max	0.18	0.10
Sodium	mg/L	≤200 (AO)	max	31	17
Sulphate	mg/L	≤500 (AO)	max	24	15
Nutrients					
Nitrate, as N, calculated	mg-N/L	10	max	3.2	2.6
Nitrite, as N	mg/-NL	1.0	max	0.029	0.005
Total Metals					
Arsenic	µg/L	10	max	0.2	0.08
Antimony	µg/L	6	max and range in average[(c)]	*17* and 0.06 to 4.5	0.03
Barium	µg/L	1,000	max	29	17.1
Boron	µg/L	5,000	max	53	28
Cadmium[(d)]	µg/L	5	max	0.07	0.25
Chromium	µg/L	50[(e)]	max	0.3	<0.06
Hexavalent chromium	µg/L	50[(e)]	max	1	-
Copper	µg/L	≤1,000 (AO)	max	0.8	0.7
Iron	µg/L	≤300 (AO)	max	19	168
Lead	µg/L	10	max	0.07	0.43
Manganese	µg/L	≤50 (AO)	max	39	4
Mercury	µg/L	1	max	0.001	<0.02
Selenium	µg/L	10	max	0.04	<0.04
Uranium	µg/L	20	max	0.20	0.05
Zinc	µg/L	≤5,000 (AO)	max	9	16
Organics-Volatiles					
Benzene	mg/L	0.005	max	<0.0005	-
Ethylbenzene	mg/L	0.0024	max	<0.0005	-
Toluene	mg/L	0.0024	max	<0.0005	-
Xylene	mg/L	0.3	max	<0.00071	-

Table 3-24 Comparison of Snap Lake Water Quality to Canadian Drinking Water Quality Guidelines

Parameter	Units	Canadian Drinking Water[a]	Type	Observed Data[b]	
				Snap Lake	SNP 02-15
Microbiology					
E. coli	CFU/100 mL	0	max	-	<1
E. coli	MPN/100 mL	0	max and range in average[c]	*1* and <1 to <1	<1
Total Coliforms	CFU/100 mL	0	max	-	*10*

Note: AO = Aesthetic objectives. Aesthetic effects (e.g., taste, odour) are taken into account when these play a role in determining whether consumers will consider the water drinkable.

SNP 02-15 = water intake from Snap Lake.

(a) Canadian drinking water guidelines are obtained from Health Canada (2012). Unless stated, the guideline concentrations are Maximum Acceptable Concentrations (MAC)

(b) Observed concentrations within the 2012 reporting period, which is January 1, 2012 to September 30, 2012 for Snap Lake and November 1, 2012 to October 31, 2012 for SNP 02-15. *Italicized* values are above the relevant MAC.

(g) Minimum and maximum average concentration calculated by area, excluding northwest arm stations.

(d) Cadmium was analyzed by Alberta Innovates Technology Futures and ALS Laboratory Group for Snap Lake and SNP 02-15, respectively

(e) Although the guideline is protective of health effects from chromium (VI), it applies to total chromium including both chromium (III) and (VI).

SNP = Surveillance Network Program; N = nitrogen; - = not available; <= less than; ≤ = less than or equal to; max = maximum. *E. coli* = *Escherichia* coli; µg/L = microgram per litre; mg/L = milligram per litre; cfu/100 mL = colony forming units per 100 millilitres; MPN/100 mL = most probable number per 100 millilitres.

Table 3-25 Summary of Temporal Trends for Total Antimony 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 Antimony (as a two-sided trend)	Diffuser (SNP02-20e)	Bottom	45	1.407	0.159	-
		Mid	45	0.424	0.671	-
		Surface	44	0.733	0.463	-
	Near-field (SNAP05)	Mid	28	1.869	0.062	-
	Mid-field (SNAP09)	Mid	29	1.441	0.150	-
	Far-field (SNAP08)	Mid	36	-0.107	0.915	-
	Northwest Arm (SNAP02A)	Mid	24	0.993	0.321	-

Note: The Seasonal Kendall Test was run using SYSTAT 13.00.01 (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.

- = no significant increasing or decreasing trend;% = percent; n = sample count.

3.4.7.1 Summary of Key Question 6

Concentrations of most water quality parameters in Snap Lake were below drinking water guidelines, with the exception of *E. coli*, total coliforms, and the metalloid antimony (total) near the diffuser under ice. Drinking water at the Mine is filtered and chlorinated prior to consumption, so drinking water at the Snap Lake camp was acceptable from a microbiological perspective (i.e., *E. coli* and coliforms). The antimony results were suspect due to indications of either contamination or analytical interference. In any case, antimony concentrations near the water intake were well below the drinking WQG. As well, raw and potable water at the Mine have been tested and reported to the local Health Authority. Antimony concentrations have been well below the levels of potential concern in those samples. Therefore, the camp workers who drink the water are not at risk. Aside from the locations near the diffuser under ice, Snap Lake water is safe for humans (pending disinfection) and wildlife to drink. Antimony will continue to be investigated through follow-up QA/QC measures. Drinking water at the Mine will continue to be tested regularly and results reported to the local Health Authority.

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?

The volume of daily discharge to Snap Lake has increased since 2004. While no clear increasing or decreasing trend in concentrations was observed for many signature parameters, loadings to Snap Lake have increased due to increases in daily discharge rates. In 2012, the annual treated effluent volume was approximately 12% of the volume of Snap Lake.

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 eight metals (barium, boron, lithium, 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 most parameters in 2012. Exceptions were two TSS results and one total aluminum concentration.

Flow-weighted average concentrations of sulphate have routinely been above the maximum average annual concentration predicted in the EAR. The CCME currently does not provide WQGs for sulphate. Because sulphate is a component of TDS (i.e., approximately 9%), it is implicitly considered as part of the ongoing aquatic toxicity testing being conducted to develop an appropriate site-specific, effects-based TDS water quality benchmark.

The 2012 TP loading to Snap Lake from the WTP was 67 kg/y, which was well below the Water License limit of 256 kg/y.

The 2012 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 2012.

Chronic toxicity was predicted to occur in treated effluent in the EAR (De Beers 2002). In 2012, one treated effluent sample from the permanent WTP showed evidence of chronic toxicity in terms of *Ceriodaphnia dubia* survival but not 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.

3.5.2 Key Question 2: Are concentrations of key water quality parameters in Snap Lake below AEMP benchmarks and Water Licence limits?

The 2012 water quality data from Snap Lake were below AEMP benchmarks and Water Licence limits, with the exception of chloride, fluoride, and nitrate. Similar to 2011, the majority of water samples collected in 2012 contained fluoride at concentrations above the CCME WQG of 0.12 mg/L, which is a conservative concentration as it includes a relatively high safety factor (CCME 2002). The maximum fluoride concentration was 0.18 mg/L. In 2012, chloride concentrations in Snap Lake were typically below the CCME WQG of 120 mg/L for chloride with the exception of one chloride result of 121 mg/L collected from the diffuser area. Approximately 3% of the 2012 nitrate samples collected in Snap Lake were above the CCME WQG for nitrate of 2.93 mg-N/L, with a maximum concentration of 3.22 mg-N/L. Average nitrate concentrations in the different lake areas remained below the CCME WQG for nitrate.

Because the primary source of fluoride, chloride, and nitrate is the treated effluent, increases in these parameters are associated with elevated calcium and hardness, which are expected to reduce the potential for toxicity effects associated with fluoride, chloride and nitrate. Proposed site-specific benchmarks and management actions for nitrate and TDS (which includes chloride and fluoride) are currently under development for the Snap Lake Mine as part of the Nitrogen and TDS Response Plans, respectively.

Whole-lake average and maximum concentrations of TDS in Snap Lake were below the Water License limit of 350 mg/L in 2012. The whole lake average concentration ranged from 187 mg/L in July, to 234 mg/L in April, with a maximum TDS concentration of 279 mg/L.

In 2012, DO concentrations in Snap Lake were considered healthy for fish and other aquatic organisms, with the exception of four locations, where field DO readings dropped below the

CCME WQG of 6.5 mg/L. At 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. Low DO concentrations near the bottom of the lake were observed during ice-covered conditions under baseline conditions. Overall, DO concentrations in Snap Lake do not appear to have decreased as a result of treated effluent discharge. In 2012, increases rather than reductions in bottom DO concentrations were observed around the diffuser relative to Northeast Lake.

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 2012, the following parameters increased above the Snap Lake normal range (i.e., baseline mean ± two standard deviations) and reference lake (Northeast Lake and Lake 13) concentrations in at least one area of Snap Lake:

- TDS, total alkalinity, 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).

Whole-lake average TDS concentrations in Snap Lake in 2012 were higher than 2011 Water Licence Renewal Application model predictions. The divergence was due to higher than predicted TDS loadings from treated effluent discharge between 2006 and 2012. The rate of increase in TDS concentrations in the lake was directly related to the loadings from the treated effluent discharge.

Measured whole-lake average concentrations of nitrate in Snap Lake have been increasing since 2005, consistent with EAR and recent modelling predictions. In 2012, concentrations of nitrate were below maximum EAR predictions.

In 2012, increases in surface and bottom DO concentrations were measured over the winter in the main basin of Snap Lake. The increase in bottom DO concentrations during ice-covered conditions near the diffuser resulted from the release of oxygenated treated effluent from the diffuser near the lake bottom.

Toxicity test results from three diffuser samples collected in April and three collected in September showed no adverse effects for any test endpoints. Algal growth was stimulated in all samples, however, with the degree of stimulation increasing at higher sample concentrations (Appendix A5).

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 2012 can be explained by the discharge of treated effluent and seasonal differences in mixing conditions in Snap 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 that were directly associated with treated effluent discharge (conductivity, most major ions, nitrogen-nutrient [e.g., nitrate, nitrite, ammonia], and eight metals [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 2012 compared to gradients observed in the first four years of minewater discharges to Snap Lake (i.e., 2004 and 2007). The increase in concentrations has now become widespread throughout the main body of Snap Lake.

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 at Northeast Lake. Higher concentrations closer to the northwest arm’s narrow connection to the main basin were evident again in 2012. Manganese concentrations were notably elevated in the northwest arm during ice-covered conditions, which may be related to prevalent lower DO measured during ice-covered and open-water periods at sampling locations in this area of Snap Lake.

Seasonal differences between ice-covered and open-water conditions were less prominent in 2012 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.

Vertical patterns in field conductivity in 2012 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. Vertical gradients in DO and water temperature were observed, typically during ice-covered conditions, related primarily to natural lake processes.

In the EAR, parameter concentrations associated with the treated effluent discharge were conservatively predicted to reach background concentrations within 44 km of Snap Lake, assuming maximum concentrations during operations. Concentrations of TDS were at background levels near King Lake, which is 25 km downstream. Results from the Downstream Lakes Special Study (Section 12.2) showed evidence of the treated effluent throughout lakes DSL1 and DSL2, and near the inlet of Lac Capot Blanc in 2012. Concentrations of Mine-related constituents reached background concentrations approximately 6 km downstream of Snap Lake.

3.5.5 Key Question 5: Is there evidence of acidification effects from the Mine on nearby waterbodies?

Based on the 2012 data, there was no 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. However, water quality in these lakes may be changing over time. These changes are particularly noticeable at IL5, where concentrations of sulphate, base cations, pH, and alkalinity are elevated compared to baseline.

The ranges in pH, total alkalinity, sulphate, and base cations in Streams S1 and S27 in 2012 were similar to baseline values; therefore, no evidence of acidification, or any spring acid pulse, was discernible in Streams S1 and S27 in 2012.

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 drinking water guidelines, with the exception of *E. coli*, total coliforms, and the metalloid antimony (total) near the diffuser under ice. Drinking water at the Mine is filtered and chlorinated prior to consumption, so drinking water at the Snap Lake camp was acceptable from a microbiological perspective (i.e., *E. coli* and coliforms). The antimony results were suspect due to indications of either contamination or analytical interference. In addition, antimony concentrations in the treated effluent have been low. In any case, antimony concentrations near the water intake were well below the drinking WQG. As well, raw and potable water at the Mine have been tested and reported to the local Health Authority. Antimony concentrations have been well below the levels of potential concern in those samples. Therefore, the camp workers who drink the water are not at risk. Aside from the locations near the diffuser under ice, Snap Lake water is safe for humans (pending disinfection) and wildlife to drink. Antimony will continue to be investigated through follow-up QA/QC measures. Drinking water at the Mine will continue to be tested regularly and results will be reported to the local Health Authority.

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 focus on investigating potential contamination, variability between samples, and alleviating hold time issues. These recommendations include 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 investigate the accuracy and precision of analyzing individual components of TP by the analytical laboratories currently used in the AEMP program (Section 12.4). Continued split sampling, which is part of the regular QA/QC procedures in the AEMP, is recommended to provide an external check of the primary laboratories completing the analyses. A limited number of nutrient spike samples should routinely be sent to several laboratories as an on-going and independent check of the accuracy of nutrient results.

Water Quality Data Interpretation:

- Review the application of the CCME fluoride, chloride, and nitrate WQGs because there are known ameliorating factors that would apply in Snap Lake. Proposed site-specific benchmarks and management actions for nitrate and TDS (which includes chloride and fluoride) are currently under development for the Snap Lake Mine as part of the Nitrogen and TDS Response Plans, respectively. In accordance with the Water Licence (MVLWB 2012), these plans will include a description of the sources of nitrogen and TDS, a description of the ecological implications of nitrogen and TDS loadings on the receiving environment, and a discussion on options for reducing loadings.
- Give consideration to parameters with concentrations that have increased beyond the normal range in Snap Lake, but for which there are no relevant AEMP benchmarks (i.e., barium, lithium, rubidium, and strontium). A site-specific benchmark for strontium is being prepared; however, it is recommended that for the remaining parameters (total barium, lithium, and rubidium), available toxicological literature be reviewed to determine the implications of increases in these parameters.

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 related to increased loadings.
- Sulphate was not identified as a key parameter during the most recent lake model update (De Beers 2011b) because the CCME do not currently provide WQGs for sulphate. Sulphate will be included in future lake model updates, because flow-weighted concentrations in the treated effluent were above EAR predictions.

- Concentrations of antimony should continue to be investigated through the follow-up QA/QC measures listed above. If the QA/QC investigation indicates that the observed values above the MAC are real, total antimony should be added to the modelling parameter suite to investigate whether any physical or chemical processes are influencing antimony concentrations and what the maximum concentrations throughout the lake are predicted to be throughout Mine operations.
- Re-visit the acidification assessment completed in 2009 using updated air modelling information and water quality data to determine whether the results from the 2009 assessment remain valid.

Study Design:

- After completion of the acidification re-assessment, the water quality sampling program in the inland lakes should be re-visited to determine whether the current design is appropriate.
- Shift the focus from spatial and seasonal trends in Snap Lake to changes downstream of Snap Lake. As the overall water quality begins to change in Snap Lake, the seasonal and spatial differences in water quality in the lake become less relevant and the temporal changes in Snap Lake and changes downstream of Snap Lake become more relevant. In response to the changes in water quality, the number of monitoring stations in Snap Lake should be reduced; information gathered from the Downstream Lakes Special Study should be used to establish new downstream AEMP stations in addition to the current KING01 station.
- Because the results of nutrient samples collected at different depths demonstrate that nutrient concentrations, particularly total phosphorus, may vary with sampling depth, reduction to a single combined sampling depth for water quality and plankton components is not recommended at this time (Section 12.4). Additional nutrient samples should be collected at mid-depth and in the euphotic zone to better define which forms of nutrients differ with sample depth and the degree to which this difference may affect other nutrient-related components and activities at Snap Lake, such as benthic invertebrates and water quality modelling. The results should be reported jointly as part of an “eutrophication indicators section” in the AEMP report.

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4 SEDIMENT QUALITY

4.1 INTRODUCTION

4.1.1 Background

This section presents sediment quality data collected from Snap Lake and Northeast Lake in 2012, as well as data collected from the provisional second reference lake, Lake 13. This section provides results of comparisons to sediment quality guidelines (SQGs), and the results of analyses of spatial and temporal patterns of target parameter concentrations. It provides the results of statistical tests evaluating the relationships between Snap Lake sediment parameters and exposure to treated effluent, as represented by conductivity at the lake bottom, and between sediment parameter concentrations in Snap Lake and two reference lakes. Among-year trends in mean concentrations of sediment quality parameters in Snap Lake and one of two reference lakes were also evaluated and are reported herein.

Sediment quality has been monitored as part of the Aquatic Effects Monitoring Program (AEMP) since 2004. The sediment quality monitoring program underwent a major change 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. 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 current mid-field 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, which is being investigated as a potential 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; Szeicz and MacDonald 2001; Vardy et al. 1997; Wolfe et al. 1996). 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.

In 2012, a separate sampling trial was undertaken to determine whether the depth of sediment sampled could be further reduced and whether this would reflect a difference in sediment chemistry results. This was a continuation of a similar trial performed in 2011, but used different sampling equipment. For the 2012 trial, sediments were collected at three stations (SNP 02-20e, SNAP03, and SNAP17) 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.

4.1.2 Objectives

The overall objective of the sediment quality monitoring program is to confirm that sediment quality in Snap Lake remains acceptable such that a healthy benthic invertebrate community is maintained. The specific objectives of the sediment quality monitoring program were:

- to characterize and interpret bottom sediment quality in Snap Lake and Northeast Lake in 2012, and make comparisons to previous years;
- to evaluate the suitability of Lake 13 as a second reference lake;
- 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[7] (MVLWB 2004, 2012).

Analysis of the 2012 sediment quality data addressed 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 lake 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 have been 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.

The sampling designs of the sediment quality and benthic invertebrate components of the monitoring program have changed over time to reflect temporal variation in exposure of the lake

[7] The Mine operated under Water Licence MV2001L2-0002 (MVLWB 2004) from June 2004 to June 2012. The Water Licence was renewed in 2012, and the Mine now operates under Water Licence MV2011L2-0004 (MVLWB 2012).

bottom to treated effluent. Additional detail regarding the Snap Lake area designations and their rationale is provided in Section 2.2.

Baseline sediment sampling was performed at 12 Snap Lake stations in 2004. Modifications were made to the 2004 sampling design in 2005 and 2006, increasing the number of stations from 12 to 18 to provide broader spatial coverage of the lake, and adjusting water depth of some locations for benthic invertebrate sampling. Snap Lake stations are shown in Table 4-1 and Figure 4-1. These 18 Snap Lake stations have been sampled annually since 2006, except as noted below:

- In 2007, the SNAP15 station could not be sampled due to unsafe conditions. This station was successfully sampled between 2008 and 2012.
- In 2007, the locations of two stations were permanently adjusted to accommodate the May 2006 commissioning of the permanent diffuser at a slightly different location from the temporary diffuser. The temporary diffuser station SNP 02-20b was renamed SNAP26 and re-designated to the near-field area. The near-field station SNAP13 was moved 70 m to the northwest, renamed SNP 02-20e, and re-designated to the diffuser area.

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). Five stations were sampled in Lake 13 in 2012 (Figure 4-3), to evaluation the potential suitability of Lake 13 as a second reference lake.

4.2.1.2 Timing of Sampling

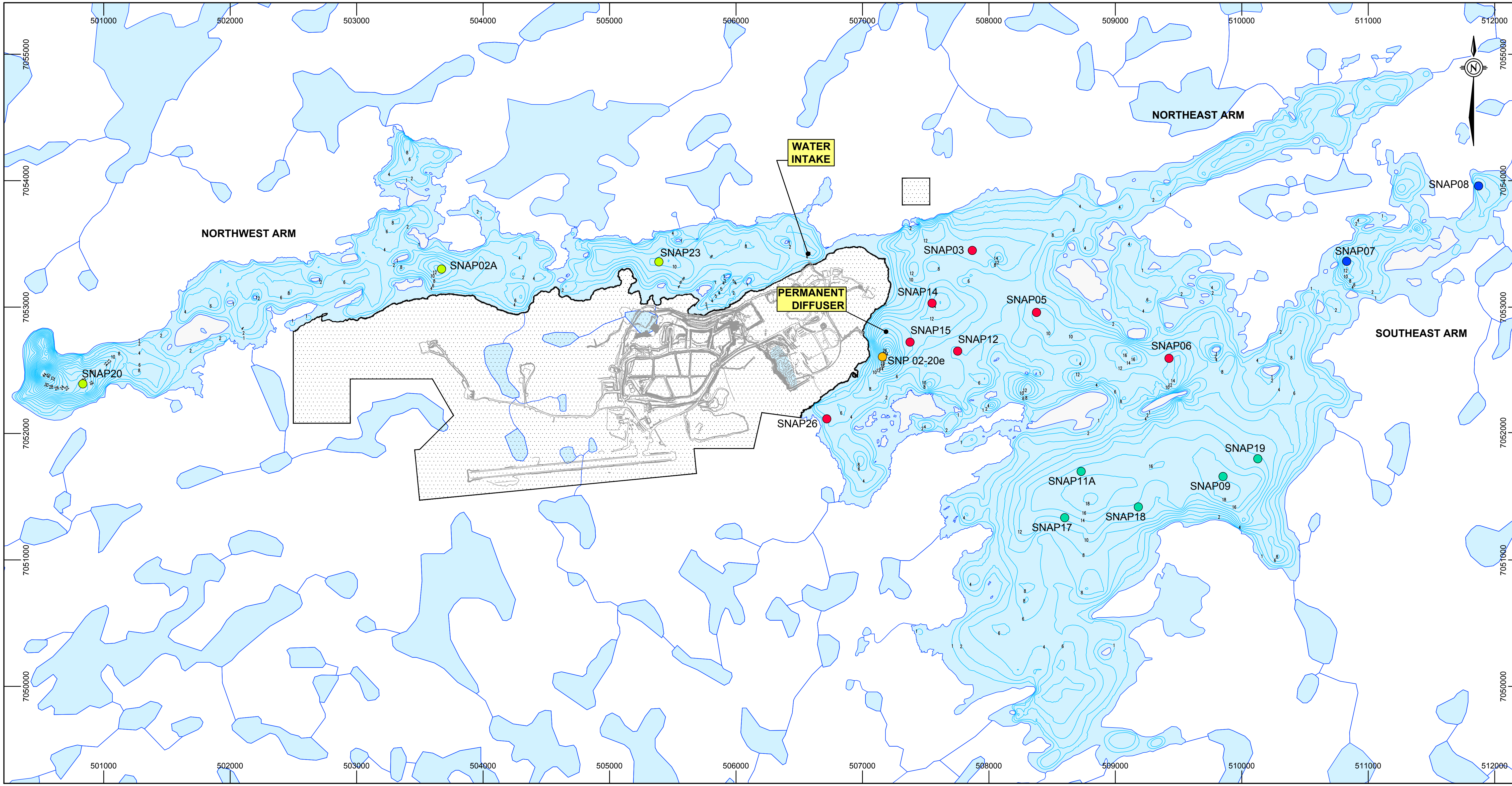
In 2012, sediment samples were collected during late summer, when ice-cover was absent on the lakes. Treated effluent was discharging through the permanent diffuser at the time of sampling. Three separate sediment sampling programs were conducted in 2012:

- Sediment depth comparison: sediment samples collected August 14, 2012, from three Snap Lake stations (SNP 02-20e, SNAP03, and SNAP17) to compare sediment quality in samples collected from the top 5 cm layer of sediment using an Ekman grab with samples collected from the top 2 cm layer of sediment using a Tech-Ops corer;
- Lake 13 reference lake sampling: sediment samples collected August 18 to 20, 2012, from five stations in Lake 13 to evaluate the suitability of Lake 13 as a potential second reference lake for the AEMP; and,
- 2012 AEMP sediment sampling: sediment samples collected September 5 to 11, 2012, from 18 stations in Snap Lake and five stations in Northeast Lake for the annual AEMP sediment quality component.

Table 4-1 Sediment Quality Stations Sampled in Snap Lake Since 2004

Lake Area	Station	2004	2005	2006	2007	2008 to 2012
Diffuser	SNP 02-20a	–	X	–	–	–
	SNP 02-20b	–	X	X	–	–
	SNP 02-20e	–	–	–	X	X
Near-field	SNAP03	X	X	X	X	X
	SNAP05	X	X	X	X	X
	SNAP06	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
	SNAP16	–	X	–	–	–
	SNAP26	–	–	–	X	X
Mid-field	SNAP09	X	X	X	X	X
	SNAP11 / 11A	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
	SNAP08	X	X	X	X	X
	SNAP10	–	X	–	–	–
Northwest Arm	SNAP01	X	X	–	–	–
	SNAP02 / 02A	X	X	X	X	X
	SNAP20	–	–	X	X	X
	SNAP23	–	–	X	X	X

X = station was sampled; – = station was not sampled.



LEGEND

- SEDIMENT QUALITY MONITORING STATION: DIFFUSER
- SEDIMENT QUALITY MONITORING STATION: NEAR-FIELD
- SEDIMENT QUALITY MONITORING STATION: MID-FIELD
- SEDIMENT QUALITY MONITORING STATION: FAR-FIELD
- SEDIMENT QUALITY MONITORING STATION: NORTHWEST ARM
- SNAP LAKE MINE FOOTPRINT
- WATERBODY
- DEPTH CONTOUR (m)

REFERENCES

DIGITAL MAP FROM MACKAY LAKE, NORTHWEST TERRITORIES, PRODUCED BY DEPARTMENT OF ENERGY, MINES AND RESOURCES. MAP 75M, ORIGINAL SCALE 1:250,000, NAD 83 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. ALL DEPTH VALUES ARE SHOWN IN METRES
2. MAP MOVED TO NAD83 COORDINATE SYSTEM

PROJECT
DE BEERS GROUP OF COMPANIES

TITLE
SEDIMENT QUALITY MONITORING STATIONS IN SNAP LAKE, 2012

Golder Associates

PROJECT	13-1349-0001.4400		FILE No.	131349000144000A4-1
DESIGN	TD	2013-03-13	SCALE AS SHOWN	REV. 0
CADD	JEF	2013-03-30		
CHECK	LMB	2013-04-19		
REVIEW	PMC	2013-04-19		

FIGURE 4-1

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LEGEND

SEDIMENT QUALITY AEMP MONITORING STATION

DEPTH CONTOUR (m)

WATERBODY

REFERENCES

DIGITIZED FROM NTS TOPOGRAPHIC MAP 75 M/10 © 1985 HER MAJESTY THE QUEEN IN RIGHT OF CANADA. DEPARTMENT OF ENERGY, MINES AND RESOURCES. PROJECTION : TRANSVERSE MERCATOR, DATUM : NAD27, COORDINATE SYSTEM : UTM ZONE 12.

REFERENCE 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.

500 0 500

SCALE METRES

PROJECT

DE BEERS GROUP OF COMPANIES

TITLE

SEDIMENT QUALITY MONITORING STATIONS IN NORTHEAST LAKE, 2012

PROJECT	13-1349-0001.4400	FILE No.	13134900014400A4-2
DESIGN	TD	2013-03-13	SCALE AS SHOWN REV. 0
CADD	JEF	2013-03-30	
CHECK	LMB	2013-04-19	
REVIEW	PMC	2013-04-19	

Golder Associates

FIGURE: 4-2

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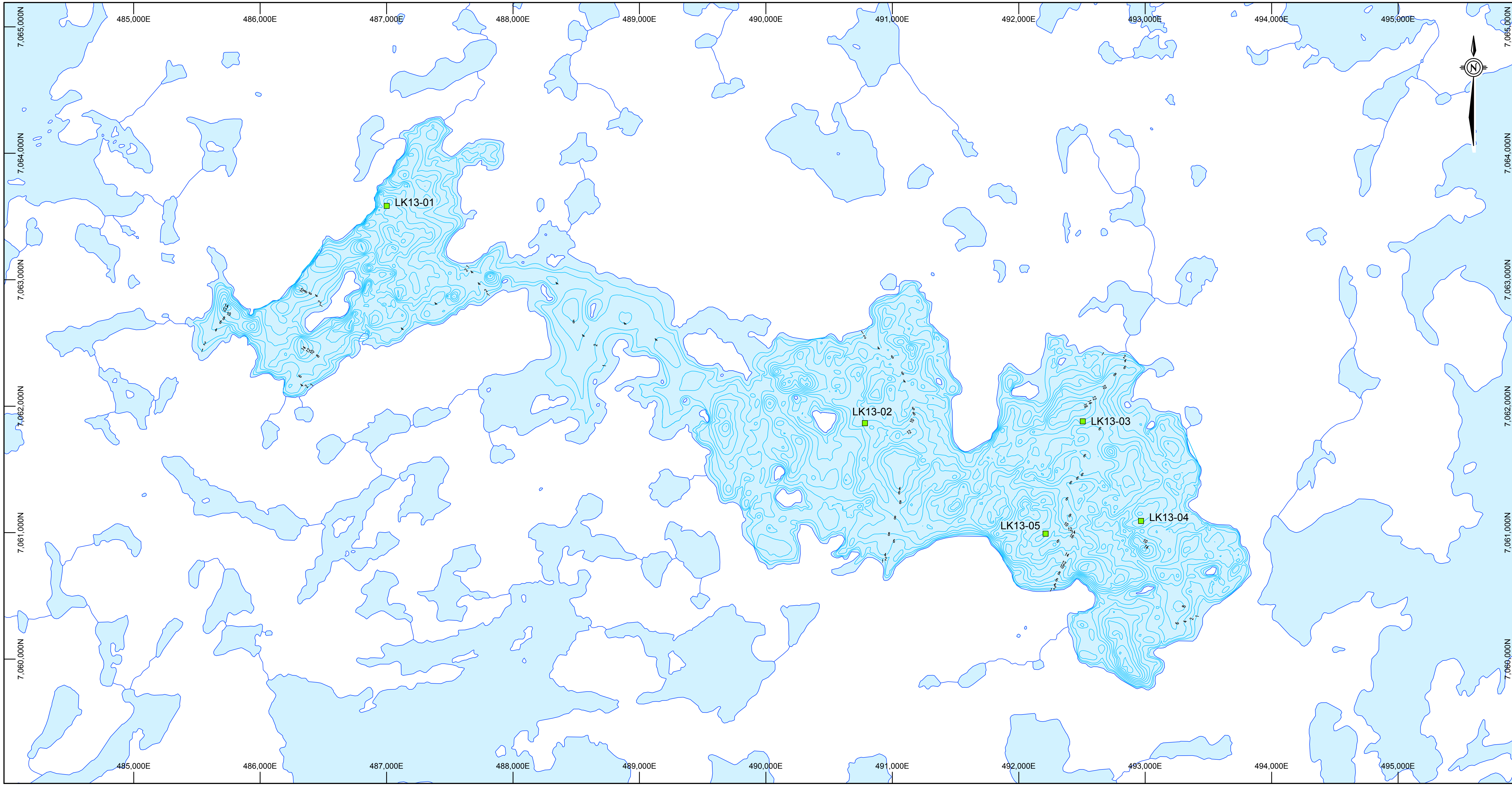

500 0 500
SCALE METRES

LEGEND

- SEDIMENT QUALITY MONITORING STATION
- WATERBODY
- DEPTH CONTOUR (m)

REFERENCE

BASE VECTOR DIGITAL CANVEC 1:50K DATA OBTAINED FROM GEOGRATIS ONLINE.

DATUM: NAD83 PROJECTION: UTM ZONE 12

PROJECT
DE BEERS
GROUP OF COMPANIES

TITLE
SEDIMENT QUALITY MONITORING STATIONS IN PROVISIONAL REFERENCE LAKE 13, 2012

Golder Associates

PROJECT	13-1349-0001.4400		FILE No.	13134900014400A4-3
DESIGN	TD	2013-03-13	SCALE AS SHOWN	REV. 0
CADD	JEF	2013-03-30		
CHECK	LMB	2013-04-19	FIGURE: 4-3	
REVIEW	PMC	2013-04-19		

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4.2.1.3 Sampling Methods

Sediment sampling stations in Snap Lake were accessed by boat in 2012. A helicopter was used to transport the boat and field crew to Northeast Lake and to Lake 13.

For all three sampling programs, three sediment grabs were collected at each station 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.

Field duplicate samples were collected at two stations during the 2012 AEMP sediment sampling program, using separately collected sets of three Ekman grab samples to sample the top 5 cm of sediment. The field duplicate stations were SNP 02-20e and NEL01.

For the sediment depth comparison study at three stations, three Ekman grabs were collected at each station and processed as described above to generate a top 5 cm composite sediment sample for each station. To sample the top 2 cm layer of sediment at these stations, a 10 cm Tech-Ops sediment corer was used. Three core samples were collected at each 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 top 2 cm composite sediment sample for each of the three stations, and transferred to sample containers for delivery to the analytical laboratory.

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) in Edmonton, Alberta, for analyses of particle size, nutrients, carbon content and total metals (Table 4-2). The suite of elements reported in the ALS total metals analysis includes a number of metalloids such as arsenic and non-metals such as selenium, which are collectively referred to as “metals” in this report.

Analyses for total metals were performed by the ALS Edmonton laboratory, and analyses for particle size, carbon content, and nutrients were performed by the ALS Saskatoon laboratory. In addition to the selenium analyses performed by the ALS Edmonton laboratory as part of their total metals package, additional analyses to compare a second analytical methodology for selenium were performed by the ALS Burnaby laboratory (Section 4.3.2).

4.2.1.5 Supporting Environmental Variables

Supporting environmental information recorded during the 2012 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 2.2.

Table 4-2 2012 Sediment Quality Parameter List for Samples Collected in Snap Lake, Northeast Lake, and Lake 13

Parameter Group	Parameter [Units]
Physical	moisture (%)
	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)]

Table 4-2 2012 Sediment Quality Parameter List for Samples Collected in Snap Lake, Northeast Lake, and Lake 13

Parameter Group	Parameter [Units]
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) [ICP-MS]
	selenium (mg/kg dw) [CCMS]
	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)

% = percent;% dw = percent dry weight; mg/kg dw = milligrams per kilogram dry weight; CCMS = collision cell inductively coupled plasma mass spectrometry; ICP-MS = inductively coupled plasma mass spectrometry; N = nitrogen; P = phosphorus; S = sulphur.

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. Specific details relevant to data analysis methods to address each key question 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 guidelines

4.2.2.2 Data Compilation and Summary

The 2012 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. For Snap Lake only, similar summary statistics were calculated for each of the five lake areas: northwest arm; diffuser; near-field; mid-field; and, far-field. These five lake areas have been used since 2005 for assessment of spatial gradients in sediment quality within Snap Lake. For 2012, summary statistics were calculated for an additional Snap Lake area, referred to hereafter as the “main basin”, which consisted of the 14 near-field, mid-field, and far-field stations. Results for the main basin were used to represent overall conditions in Snap Lake, and were used in the weight-of-evidence assessment for the qualitative integration of all the AEMP components (Section 13). Northwest arm stations were excluded because of the consistent occurrence of anomalously high concentrations of some sediment quality parameters unrelated to the Mine, and the diffuser station was excluded because elevated parameter concentrations related to the Mine, if present, would be expected to occur at that station and therefore the diffuser station was assessed separately from the main basin.

Statistical analyses were performed using SYSTAT 13 (SYSTAT Software Inc. 2009). Concentrations reported as less than their detection limit (DL) were replaced with values equal to half their DL prior to statistical analyses. Parameters that were undetected in sediments from at least 90 percent (%) of the sampling stations would have been screened out from the statistical analyses; none were excluded in 2012. When screening data against this criterion, data were reviewed to confirm that concentrations above DLs did not occur exclusively at stations near the diffuser.

The top 5 cm and top 2 cm sediment quality data from the stations sampled for the sediment depth comparison were compared by calculating the relative percent difference (RPD) between the concentrations in the two sample types:

$$RPD = (\text{top 5 cm} - \text{top 2 cm}) / [(\text{top 5 cm} + \text{top 2 cm})/2] \times 100$$

The RPD is the same formula used to compare results from field or laboratory duplicate analyses, and is a measure of analytical precision. RPDs were calculated for each parameter for the three stations for which both sediment depths were sampled. 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 seven metals analyzed in Snap Lake, Northeast Lake, and 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.

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

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 assessed using two approaches:

- testing for a spatial gradient associated with bottom conductivity as an indicator of treated effluent exposure; and,
- testing for statistically significant differences in mean concentrations of sediment quality parameters between the Snap Lake main basin and reference lakes.

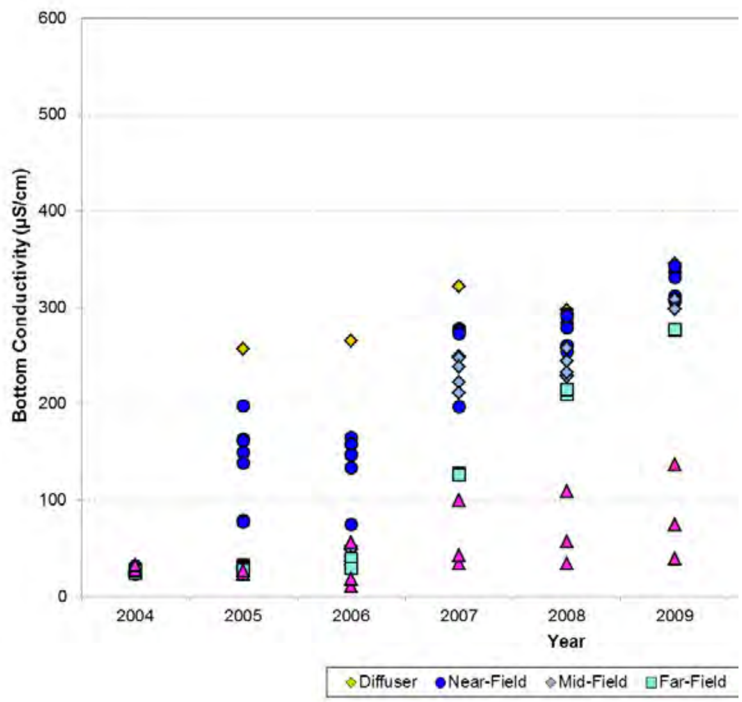
Conductivity Gradient

Conductivity measured at the lake bottom was used as the indicator of exposure to treated effluent. Treated effluent released during the ice-covered winter months sinks, and the plume spreads out along the bottom of Snap Lake. Because the lake bottom topography is complex, exposure to the treated effluent is not closely related to distance from the diffuser. For instance, exposure is greater at deeper stations compared to shallower stations. Bottom conductivity measurements account for both the complex bottom topography of Snap Lake and for water depth variation among stations.

Changes in the spatial distribution of bottom conductivity in Snap Lake during late-winter sampling conducted in approximately April from 2004 to 2012 are illustrated in Figure 4-4. April data were only available for those sediment stations that were also sampled as part of the water quality component. Although sediment sampling has been conducted in September since 2009, the late-winter conductivity data are provided because treated effluent concentrations are highest and spatial gradients are strongest at that time. September bottom conductivity data are less variable due to the mixing of lake water that occurs during the summer open-water period.

Discharge of treated effluent began in June 2004 through a temporary diffuser, and has occurred from a permanent diffuser since May 2006. The gradients of treated effluent exposure present in Snap Lake in 2005 and 2006 are shown in Figure 4-4. In those years, bottom conductivity was a good tracer because of the strong separation between the diffuser and far-field stations, but over time this gradient has diminished as ongoing discharge has resulted in increased overall conductivity throughout Snap Lake, except for the northwest arm which is less influenced by this discharge. Although bottom conductivity is no longer a good tracer of the treated effluent influence, the underlying hydrology and mixing of treated effluent dispersion within Snap Lake have not changed. Therefore, use of conductivity to assess spatial trends in sediment parameter concentrations is still appropriate.

Figure 4-4 Late-Winter Bottom Conductivity in Snap Lake, April 2004 to April 2012



µS/cm= microSiemens per centimetre.

Least squares linear regression analyses were used to determine whether parameter concentrations in Snap Lake sediments were significantly related to exposure to the treated effluent expressed as bottom conductivity. The 2006 bottom conductivity data[8] were used for these analyses because of the strength of the gradient. Although the 2005 data exhibited a similar gradient, they were not used because only 11 of the current 18 Snap Lake stations were sampled. One near-field station (SNAP26) was excluded from these analyses because corresponding conductivity data were not available. In cases where outliers were identified during the regression analyses, the regressions were repeated after removing those data. If removal of data resulted in a change in statistical significance, the results of both analyses were reported.

The regression results (probability [*P*]) were considered significant at $P<0.1$, consistent with the recommendations of Environment Canada (2012) for aquatic Environmental Effects Monitoring (EEM). A regression equation with a positive slope that was significantly different from zero was interpreted as an indication of a potential effect on sediment quality from exposure to the treated effluent discharge (Section 4.4.4). For parameters designated as potentially affected based on this analysis, effects were further analyzed by statistically comparing the slope of the 2012 regression line with the slope based on 2004 (baseline) data, using bottom conductivity as the independent variable. SYSTAT's general linear model procedure was used to evaluate

[8] The 2006 bottom conductivity data have been used for these analyses since 2009. This differs from procedures used in 2007 and 2008 in which the current-year bottom conductivity data were used.

differences between these slopes. This additional analysis was performed to control for any pre-existing spatial trend in sediment quality, which could be erroneously interpreted as a Mine-related effect.

Among-Area Comparisons

Statistical comparisons were made between the Snap Lake main basin, the Northeast Lake reference lake, and the provisional second reference lake, Lake 13, using mean sediment parameter concentrations.

Statistical analyses were performed using a one-way analysis of variance (ANOVA) or the Kruskal-Wallis test (Sokal and Rohlf 1995), which is the non-parametric equivalent of ANOVA. Before statistical testing, data were tested for normality using the Shapiro-Wilk test and for homogeneity of variance using Bartlett's test. Where appropriate, data were $\log_{10}$ transformed to eliminate deviations from normality or heterogeneity of variance, or else a non-parametric test was used. The northwest arm and diffuser stations were excluded from the statistical analyses. Results of statistical tests were considered significant at $P<0.1$. After a significant ANOVA result, the following comparisons were conducted using planned orthogonal contrasts (Sokal and Rolf 1995):

- Pooled Northeast Lake and Lake 13 compared to Snap Lake main basin;
- Northeast Lake compared to Lake 13; and,
- Northeast Lake compared to Snap Lake main basin.
- Results of contrasts were considered significant at $P<0.03$, after a Dunn-Ŝidák correction for multiple comparisons from the original $P<0.1$ (Sokal and Rolf 1995).

4.2.2.5 Evaluation of Temporal Trends

To illustrate temporal trends in sediment quality in Snap Lake, data collected from 2004 to 2012 were plotted as lake area means; this consisted of the five existing lake areas, as well as the new main basin lake area that was added in 2012. The 2008 to 2012 means for Northeast Lake, and the 2012 means for Lake 13, were also included for comparison. Although treated effluent exposure varied within each sampling area, there have been clear stepwise increases in exposure in parts of Snap Lake, from 2004 to 2005 and 2006 to 2007 in the near-field area, and from 2006 to 2007 in the mid-field and far-field areas, as reflected in the bottom conductivity data (Figure 4-4).

Statistical analyses were performed to identify statistically significant ($P<0.10$) temporal trends, using a non-parametric Mann-Kendall test (Gilbert 1987). Both increasing and decreasing temporal trends were identified, for each of the five existing areas within Snap Lake and also for the main basin. Trend analyses were also performed for Northeast Lake, as five years of monitoring data were now available.

To evaluate whether sediment quality in Snap Lake has changed relative to baseline conditions, mean concentrations were compared with baseline (2004) conditions expressed as normal ranges calculated for each parameter. For particle size, total organic carbon (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, and 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 <50 µS/cm. For each sediment quality parameter, mean concentrations calculated annually for each lake area within Snap Lake, as well as means for Northeast Lake and Lake 13, were compared to their respective normal ranges on time-series plots.

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 2012 sediment quality samples are provided in Appendix 4A.

4.3.2 Summary of Results

In 2012, qualifiers were assigned to sediment sample data for DL increases for three nutrient parameters due to interference from sample matrix effects, and for detection of a number of metals in method blanks. With one exception, these failures to meet data quality objectives were relatively minor and not expected to adversely affect data quality. The one exception was available ammonium; the laboratory DL increase was 10 times higher for the depth comparison samples than for the AEMP or Lake 13 samples and therefore limited the ability compare results between the top 2 cm and top 5 cm samples for this parameter.

A total of six method blanks were analysed, two for each sediment sampling program. Barium, copper, and molybdenum were each detected in one method blank, and boron, chromium, nickel, and zinc were detected in two method blanks. In all cases the concentrations reported for the method blanks were well below the ranges reported for the sediment samples.

The specified DLs were met for most parameters; however, DLs were not met for inorganic carbon, available ammonium, available nitrate, available phosphate, chromium, potassium, and selenium. With the exception of available ammonium as noted previously, these 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 most analyses. Holding times were not met for inorganic and organic carbon analyses performed on the AEMP and Lake 13 samples, or for moisture content analyses on the AEMP samples. 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 Lake 13 samples or for the 2 cm layer depth comparison samples.

Results from analyses of laboratory duplicates and laboratory reference materials met their respective data quality objectives.

Field duplicate samples were collected at one Snap Lake station and one Northeast Lake station in 2012, 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 generally agreement in terms of measured parameter concentrations, except for available ammonium, TKN, total nitrogen, and available sulphate in one sample, and available phosphate in both samples.

Re-analysis of a number of parameters in one or more samples was requested because the initial results were either noticeably higher or more variable than all samples in previous years. Available ammonium, available nitrate, available phosphate, and available potassium were re-analysed in the six depth comparison samples; arsenic, barium, and manganese were re-analysed in the five Lake 13 samples. The original results were confirmed in all cases.

From 2004 to 2008, sediment selenium concentrations reported for the AEMP were measured by hydride vapour atomic absorption spectrometer (HVAAS). The ALS Edmonton laboratory discontinued HVAAS analyses in 2008, and has reported sediment selenium concentrations measured by inductively coupled plasma-mass spectrometry (ICP-MS) since 2009. In anticipation of ALS Edmonton changing to collision cell inductively coupled plasma-mass spectrometry (CCMS) for selenium analyses at some point, a comparison of these three selenium methods (HVAAS, ICP-MS, and CCMS) was performed in 2010 (De Beers 2011). In 2012, sediment samples were analysed for selenium by both ICP-MS and CCMS (Appendix 4A). As of November 2012, the ALS Edmonton laboratory is using CCMS for sediment selenium analyses.

4.4 RESULTS

4.4.1 Supporting Environmental Variables

Sediment sampling depths ranged from 8.1 to 14.5 m at Snap Lake stations, except for the shallower near-field Station SNAP26 (5.8 m) and the deeper diffuser Station SNP 02-20e (26 m). Sampling depths ranged from 10.5 to 12.8 m in Northeast Lake, and from 10.3 to 14.7 m in Lake 13.

Bottom conductivity measured at Snap Lake stations in September 2012 ranged from 91 to 412 µS/cm; conductivity was lower at northwest arm stations (91 to 210 µS/cm) than at the diffuser and main basin stations (390 to 412 µS/cm). Bottom conductivity was 22 µS/cm at all Northeast Lake stations, and 19 to 20 µS/cm at all Lake 13 stations, during the August and September 2012 sediment sampling.

4.4.2 Summary of 2012 Sediment Quality Data

Results from the 2012 AEMP and Lake 13 sediment sampling programs are summarized in Section 4.4.2.1, and results from the 2012 sediment depth comparison sampling program are summarized in Section 4.4.2.2. All sediment quality data are reported on a dry weight (dw) basis, except for moisture content.

4.4.2.1 AEMP Stations and Lake 13

The 2012 raw sediment quality data for Snap Lake, Northeast Lake, and Lake 13 are provided in Appendix 4B (Table 4B-1). Whole-lake means and summary statistics for 2012 Snap Lake sediments are presented in Table 4-5; for comparison, summarized baseline sediment chemistry data collected in 1999 and 2004 are provided in the same table. Whole-lake means and summary statistics for 2012 Northeast Lake and Lake 13 sediments are presented in Table 4-6.

Snap Lake sediments were comprised primarily of fine-grained silt and clay, with smaller amounts of sand. With one exception, the percentage of fines ranged from 95% to 99%. Near-field Station SNAP26 had a higher sand content and only 75% fines, which was similar to what was observed in previous years for this station. Sediments from far-field Station SNAP08 consisted of 97% fines in 2012, which was consistent with all previous years except 2010 when sediments from this far-field station were almost entirely sand. Sediments from the Northeast Lake and Lake 13 stations were similar to Snap Lake sediments, consisting of 95% to 99% fine-grained materials.

Table 4-5 Summary of Baseline and 2012 Sediment Quality Data for Snap Lake, as Whole-Lake Means and Summary Statistics

Parameter	Units	Baseline (1999 to 2004) Bulk Sediment Samples						2012 Top 5 cm Sediment Samples					
		n	Mean	SD	Median	Minimum	Maximum	n	Mean	SD	Median	Minimum	Maximum
Particle Size													
Sand	% dw	15	25.2	32.9	8.0	3.0	80.3	18	3.82	5.58	2.41	0.69	25.6
Silt	% dw	15	30.9	8.4	29	18.9	46	18	85.1	4.9	86.2	69	91.3
Clay	% dw	15	43.8	27.3	58.0	0.8	68.0	18	11.1	2.7	11.5	5.49	15.5
Fines (silt+clay)	% dw	15	74.7	32.9	92	19.7	97	18	96.2	5.6	97.6	74.5	99.3
Carbon													
Total carbon	% dw	11	19.6	4.8	19.6	7.7	27.4	18	18.2	2.6	18.3	12.1	23.2
Inorganic carbon	% dw	15	0.10	0.12	0.05	<0.01	0.4	18	0.12	0.03	0.13	0.05	0.19
Total organic carbon	% dw	15	18.0	5.2	19.5	7.7	27.3	18	18.1	2.6	18.1	12	23.1
Nutrients													
Available ammonium, as N	mg/kg dw	-	-	-	-	-	-	18	24.8	17.9	16.9	8.6	72
Available nitrate, as N	mg/kg dw	-	-	-		-	-	18	10.5	9.6	7.3	<4.0	28.8
Total Kjeldahl nitrogen (TKN)	% dw	11	1.44	0.37	1.47	0.58	1.93	18	1.31	0.23	1.38	0.878	1.63
Total nitrogen	% dw	11	1.53	0.34	1.55	0.66	1.95	18	1.35	0.18	1.36	0.931	1.65
Available phosphate, as P	mg/kg dw	-	-	-	-	-	-	18	22.1	21.7	19.1	<4.0	98.7
Available potassium	mg/kg dw	-	-	-	-	-	-	18	119	28	112	69	182
Available sulphate, as S	mg/kg dw	-	-	-	-	-	-	18	151	107	118	32.5	509
Total Metals													
Aluminum	mg/kg dw	16	14,318	2,975	13,500	8,990	20,300	18	16,022	2,469	16,900	9,300	18,200
Antimony	mg/kg dw	16	<0.1	0.02	<0.1	<0.1	<0.2	18	0.28	0.13	0.26	0.12	0.59
Arsenic	mg/kg dw	16	2.5	1.0	2.4	0.9	4.4	18	2.00	0.95	1.78	1.2	5.33
Barium	mg/kg dw	16	183	271	107	68.6	1180	18	82.8	42.1	75.5	48.9	240
Beryllium	mg/kg dw	16	0.9	0.2	1.0	0.6	1.4	18	1.03	0.19	1.03	0.61	1.41
Bismuth	mg/kg dw	16	0.37	0.16	0.30	<0.2	0.70	18	0.69	0.13	0.72	0.37	0.9
Boron	mg/kg dw	12	13.1	5.1	10.5	7.0	22.0	18	19.5	6.0	19.6	7.2	31.2
Cadmium	mg/kg dw	16	0.68	0.16	0.65	0.50	1.10	18	0.66	0.16	0.62	0.45	1.02
Calcium	mg/kg dw	12	4,217	646	4,000	3,400	5,400	18	4,387	729	4,400	2,190	5,390
Cesium	mg/kg dw	16	1.81	0.62	1.70	1.20	3.90	18	1.85	0.34	1.87	1.29	2.49
Chromium	mg/kg dw	16	35.4	8.6	33.5	23.9	57.2	18	34.6	4.0	34.3	27.1	41.2
Cobalt	mg/kg dw	16	12.1	3.2	11.0	8.6	20.7	18	17.6	14.0	12.1	7.64	64.9
Copper	mg/kg dw	16	93.0	16.3	93.9	66.5	118	18	107	10	107	75.7	125
Iron	mg/kg dw	16	24,225	9,067	22,500	9,300	42,100	18	37,578	29,757	28,750	13,600	132,000
Lead	mg/kg dw	16	5.34	1.47	5.10	3.50	9.70	18	5.40	0.74	5.23	4.06	7.22

Table 4-5 Summary of Baseline and 2012 Sediment Quality Data for Snap Lake, as Whole-Lake Means and Summary Statistics

Parameter	Units	Baseline (1999 to 2004) Bulk Sediment Samples						2012 Top 5 cm Sediment Samples					
		n	Mean	SD	Median	Minimum	Maximum	n	Mean	SD	Median	Minimum	Maximum
Lithium	mg/kg dw	16	21.1	7.9	20.9	13	47	18	20.1	3.6	20.3	13.9	27.5
Magnesium	mg/kg dw	12	3,723	1,566	3,470	2,190	8,370	18	3,509	565	3,520	2,490	4,640
Manganese	mg/kg dw	16	288	88	265	146	434	18	596	1,577	241	129	6,910
Mercury	mg/kg dw	16	<0.05	0.021	<0.05	<0.05	0.10	18	0.038	0.018	<0.05	<0.05	0.084
Molybdenum	mg/kg dw	16	9.0	3.5	8.4	4.9	18.7	18	10.5	3.5	10.8	4.49	16.4
Nickel	mg/kg dw	16	41.0	6.7	38.9	34.8	59.4	18	40.8	6.5	39.6	32.7	64.5
Phosphorus	mg/kg dw	15	1,567	667	1,500	600	2,750	18	1,200	384	1,115	630	2,470
Potassium	mg/kg dw	12	1,742	954	1,440	990	4,590	18	1,423	355	1,375	920	2,080
Rubidium	mg/kg dw	16	13.6	5.7	12.4	9.0	33.0	18	12.6	2.8	12.6	8.5	18
Selenium	mg/kg dw	16	<2	0	<0.1	<0.1	<2	18	1.73	0.25	1.73	1.12	2.2
Silver	mg/kg dw	16	<0.2	0	<0.2	<0.2	<0.2	18	0.17	0.11	<0.2	<0.2	0.54
Sodium	mg/kg dw	12	242	51	200	200	300	18	398	95	430	150	490
Strontium	mg/kg dw	16	27.2	5.0	26.0	21.0	42.0	18	58.2	17.1	56.9	24.2	84.4
Thallium	mg/kg dw	16	0.18	0.11	0.13	0.07	0.4	18	0.15	0.05	0.14	0.08	0.285
Tin	mg/kg dw	12	<2	0	<2	<2	<2	18	2.2	2.6	<2	<2	9.5
Titanium	mg/kg dw	16	452	171	439	222	982	18	224	42	215	177	333
Uranium	mg/kg dw	16	8.28	2.78	7.86	4.53	13.1	18	8.6	2.0	9.3	5.02	13
Vanadium	mg/kg dw	16	31.2	6.5	30.6	19.3	49.8	18	32.9	3.8	33.4	25	39
Zinc	mg/kg dw	16	185	50	176	124	321	18	143	25	137	110	208

- = not applicable / not available; <= less than detection limit; n = sample size; SD = standard deviation of the mean; N = nitrogen; P = phosphorus; S = sulphur; cm = centimetre;% dw = percent dry weight; mg/kg dw = milligrams per kilogram dry weight.

Table 4-6 Summary of 2012 Sediment Quality Data for Northeast Lake and Lake 13, as Whole-Lake Means and Summary Statistics

Parameter	Units	2012 Northeast Lake Top 5 cm Sediment Samples						2012 Lake 13 Top 5 cm Sediment Samples					
		n	Mean	SD	Median	Minimum	Maximum	n	Mean	SD	Median	Minimum	Maximum
Particle Size													
Sand	% dw	5	2.58	1.53	2.9	0.89	4.48	5	3.21	1.08	3.19	1.84	4.85
Silt	% dw	5	86.2	1.8	86.9	83.2	87.8	5	79.5	2.3	78.6	77.7	83.4
Clay	% dw	5	11.2	2.0	11.5	8.77	13.9	5	17.2	3.1	18.1	11.7	19.1
Fines (silt+clay)	% dw	5	97.4	1.5	97.1	95.6	99.1	5	96.7	1.1	96.7	95.1	98.2
Carbon													
Total carbon	% dw	5	16.5	1.0	16.6	15.2	17.9	5	8.1	0.8	7.9	7.4	9.3
Inorganic carbon	% dw	5	0.10	0.05	0.11	<0.10	0.17	5	0.17	0.04	0.19	0.12	0.21
Total organic carbon	% dw	5	16.4	1.0	16.6	15.0	17.8	5	7.9	0.8	7.8	7.2	9.2
Nutrients													
Available ammonium, as N	mg/kg dw	5	65.5	24.3	68.9	37.5	99.8	5	26.1	9.7	25.4	15	41.7
Available nitrate, as N	mg/kg dw	5	4.3	4.1	<4.0	<4.0	11.5	5	2.8	0.4	<6.0	<4.0	<6.0
Total Kjeldahl nitrogen (TKN)	% dw	5	0.98	0.12	0.94	0.86	1.19	5	0.684	0.067	0.653	0.632	0.799
Total nitrogen	% dw	5	1.19	0.09	1.18	1.09	1.33	5	0.679	0.086	0.632	0.615	0.821
Available phosphate, as P	mg/kg dw	5	12.7	10.3	8.8	<2.0	27.6	5	21.4	18.0	20.9	<4.0	50.3
Available potassium	mg/kg dw	5	120	25	116	91	160	5	135	42	122	102	207
Available sulphate, as S	mg/kg dw	5	102	59	74.2	36.2	181	5	26.0	21.7	21.4	9.2	62
Total Metals													
Aluminum	mg/kg dw	5	18,580	1,936	19,000	15,400	20,100	5	15,100	1,063	15,100	13,600	16,600
Antimony	mg/kg dw	5	0.34	0.12	0.28	0.25	0.52	5	0.14	0.02	0.14	0.12	0.17
Arsenic	mg/kg dw	5	2.97	0.60	2.84	2.4	3.96	5	17.7	12.4	15.9	4.97	37.2
Barium	mg/kg dw	5	147	22	141	130	185	5	400	560	173	121	1,400
Beryllium	mg/kg dw	5	1.34	0.18	1.34	1.05	1.53	5	0.82	0.11	0.79	0.74	1.01
Bismuth	mg/kg dw	5	1.20	0.18	1.27	0.9	1.34	5	0.86	0.15	0.91	0.64	0.99
Boron	mg/kg dw	5	19.8	2.3	20.2	16.9	23.2	5	6.7	0.7	6.4	6.1	7.6
Cadmium	mg/kg dw	5	0.84	0.17	0.81	0.66	1.11	5	0.49	0.12	0.45	0.36	0.65
Calcium	mg/kg dw	5	3,540	494	3,390	3,070	4,370	5	2,362	90	2,330	2,300	2,520
Cesium	mg/kg dw	5	3.22	0.21	3.18	3.01	3.46	5	3.37	0.39	3.51	2.92	3.81
Chromium	mg/kg dw	5	55.5	3.7	55.5	49.8	59.9	5	55.7	10.0	53.6	47.3	72
Cobalt	mg/kg dw	5	20	9.7	13.7	12.4	31.4	5	34.3	13.8	33.7	14.2	52.2
Copper	mg/kg dw	5	135	12.4	135	117	152	5	64.9	6.1	66.2	54.2	69.6
Iron	mg/kg dw	5	34,580	18,979	29,000	19,100	66,800	5	57,480	23,656	57,600	27,700	93,100
Lead	mg/kg dw	5	7.38	0.84	7.54	6	8.27	5	7.85	1.86	7.69	5.81	10.4

Table 4-6 Summary of 2012 Sediment Quality Data for Northeast Lake and Lake 13, as Whole-Lake Means and Summary Statistics

Parameter	Units	2012 Northeast Lake Top 5 cm Sediment Samples						2012 Lake 13 Top 5 cm Sediment Samples					
		n	Mean	SD	Median	Minimum	Maximum	n	Mean	SD	Median	Minimum	Maximum
Lithium	mg/kg dw	5	33.3	1.9	32.9	31.3	36.1	5	35.5	4.6	35.4	30.1	40.8
Magnesium	mg/kg dw	5	5,548	236	5,500	5,280	5,830	5	5,326	694	5,310	4,490	6,030
Manganese	mg/kg dw	5	460	268	391	263	927	5	11,803	19,573	4,840	343	46,600
Mercury	mg/kg dw	5	0.038	0.018	<0.050	<0.050	0.06	5	0.033	0.018	<0.050	<0.050	0.065
Molybdenum	mg/kg dw	5	10.1	3.7	10.3	5.34	14.6	5	7.83	2.45	7.47	4.63	11.5
Nickel	mg/kg dw	5	88.8	17.2	81.1	69.2	108	5	58.6	10.4	58.8	43.3	71.8
Phosphorus	mg/kg dw	5	1,032	175	990	850	1,320	5	1,144	178	1,190	860	1,320
Potassium	mg/kg dw	5	2,870	142	2,860	2,670	3,010	5	2,864	367	2,880	2,290	3,250
Rubidium	mg/kg dw	5	22.7	1.1	22.3	21.3	24.1	5	23.2	2.7	23.5	19.4	26.2
Selenium	mg/kg dw	5	1.70	0.21	1.66	1.44	2.03	5	1.03	0.17	1.05	0.74	1.17
Silver	mg/kg dw	5	0.14	0.06	<0.20	<0.20	0.22	5	<0.20	0.00	<0.20	<0.20	<0.20
Sodium	mg/kg dw	5	150	12.2	150	130	160	5	168	8	170	160	180
Strontium	mg/kg dw	5	28.0	3.1	27.4	25.8	33.3	5	24.5	9.2	21	18.4	40.8
Thallium	mg/kg dw	5	0.197	0.046	0.199	0.149	0.266	5	0.22	0.09	0.21	0.119	0.36
Tin	mg/kg dw	5	1.6	1.3	<2	<2	3.8	5	<2	0.0	<2	<2	<2
Titanium	mg/kg dw	5	374	40	386	308	414	5	439	70	417	373	525
Uranium	mg/kg dw	5	11.7	1.1	11.9	9.9	12.7	5	7.49	2.14	6.71	6.27	11.3
Vanadium	mg/kg dw	5	42.5	3.7	42.6	36.6	45.7	5	47.2	4.0	47.9	41.4	52
Zinc	mg/kg dw	5	189	22	182	163	214	5	112	10	114	95.2	120

- = not applicable / not available; <= less than detection limit. n = sample size; SD = standard deviation of the mean; N = nitrogen; P = phosphorus; S = sulphur; cm = centimetre;% dw = percent dry weight; mg/kg dw = milligrams per kilogram dry weight.

Total organic carbon (TOC) concentrations ranged from 12% to 23% in Snap Lake sediments, with all but two samples having TOC concentrations greater than 15%. Sediments from the Northeast Lake stations had TOC concentrations between 15% and 18%. In contrast, TOC concentrations at Lake 13 stations were lower, ranging from 7% to 9%.

Mean nutrient concentrations were higher in Snap Lake sediments than in Northeast Lake and Lake 13 sediments, with the exception of available ammonium and available potassium. Mean nutrient concentrations were higher in Northeast Lake sediments than in Lake 13 sediments, except for available phosphate and available potassium.

Concentrations of most of the 41 nutrients and metals analysed in Snap Lake sediments varied by less than a factor of five among individual stations. Parameters with larger concentration ranges were available ammonium, available nitrate, available phosphate, available sulphate, cobalt, iron, manganese, selenium, and tin. Station SNAP20 at the head of the northwest arm had sediment concentrations of cobalt, iron, and manganese that were considerably higher than any other Snap Lake stations and appeared to be unrelated to treated effluent discharge.

Mean concentrations of a number of parameters were higher in Northeast Lake and/or Lake 13 sediments than in Snap Lake sediments. Parameter concentrations in Northeast Lake and Lake 13 sediments were generally less variable than in Snap Lake sediments; concentrations varied by less than a factor of four except for available nitrate, available phosphate, and available sulphate in Northeast Lake, and available phosphate, available sulphate, arsenic, barium, and manganese.

Concentrations of mercury and silver were undetected in the majority of Snap Lake and Northeast Lake sediment samples until 2010 when these parameters were detected with increasing frequency, but at concentrations close to their respective DLs. This pattern continued in 2011 and 2012, although concentrations were similar to those measured in 2010. Tin was undetected in most sediment samples until 2012, when it was detected at five Snap Lake diffuser and main basin stations and one Northeast Lake station. The highest tin concentrations were reported at mid-field and far-field Snap Lake stations, rather than at the diffuser.

Sampling area mean concentrations and summary statistics for sediments collected in Snap Lake in 2012 are presented in Table 4-7. The number of stations included in each Snap Lake sampling area varied from one to seven. The Snap Lake main basin area, consisting of the 14 near-field, mid-field, and far-field stations, was also included. To facilitate spatial comparisons, the 2012 Snap Lake area means, Northeast Lake means, and Lake 13 means are presented graphically in Appendix 4B (Figure 4B-1).

Table 4-7 Summary of Sediment Quality Data for Snap Lake, as Sampling Area Means and Summary Statistics, 2012

Parameter	Units	Northwest Arm			Diffuser		Near-Field			Mid-Field			Far-Field			Main Basin		
		n	Mean	SD	n	Mean	n	Mean	SD	n	Mean	SD	n	Mean	SD	n	Mean	SD
Particle Size																		
Sand	% dw	3	2.3	0.3	1	4.06	7	5.3	9.0	5	3.0	2.0	2	2.9	0.4	14	4.1	6.3
Silt	% dw	3	85.2	2.9	1	83.5	7	84.0	6.8	5	85.1	3.7	2	90.0	1.9	14	85.2	5.5
Clay	% dw	3	12.6	2.7	1	12.5	7	10.7	2.8	5	11.9	2.1	2	7.2	2.3	14	10.6	2.8
Fines (silt + clay)	% dw	3	97.7	0.2	1	96.0	7	94.7	8.9	5	97.0	2.0	2	97.2	0.4	14	95.9	6.3
Carbon																		
Total carbon	% dw	3	17.8	4.9	1	18.4	7	17.8	1.7	5	17.3	0.9	2	22.6	0.8	14	18.3	2.2
Inorganic carbon	% dw	3	0.14	0.04	1	0.10	7	0.12	0.03	5	0.11	0.04	2	0.14	0.01	14	0.12	0.03
Total organic carbon	% dw	3	17.6	4.9	1	18.3	7	17.6	1.7	5	17.2	0.8	2	22.5	0.8	14	18.2	2.2
Nutrients																		
Available ammonium, as N	mg/kg dw	3	34.1	8.7	1	18.1	7	20.3	11.6	5	11.4	2.5	2	63.5	12.0	14	23.3	19.6
Available nitrate, as N	mg/kg dw	3	2.0	0.0	1	3.0	7	13.5	10.8	5	14.0	9.9	2	7.8	8.1	14	12.9	9.7
Total Kjeldahl nitrogen (TKN)	% dw	3	1.20	0.28	1	1.60	7	1.31	0.27	5	1.30	0.20	2	1.39	0.0	14	1.32	0.22
Total nitrogen	% dw	3	1.33	0.34	1	1.36	7	1.32	0.13	5	1.28	0.07	2	1.63	0.03	14	1.35	0.15
Available phosphate, as P	mg/kg dw	3	12.2	7.0	1	98.7	7	15.0	8.9	5	24.8	10.0	2	16.7	20.7	14	18.7	11.1
Available potassium	mg/kg dw	3	95	26	1	112	7	137	33	5	110	9	2	119	30	14	125	28
Available sulphate, as S	mg/kg dw	3	133	90	1	102	7	200	146	5	92.8	25.3	2	179	103	14	158	116
Total Metals																		
Aluminum	mg/kg dw	3	17,433	862	1	15,400	7	15,586	2,903	5	17,500	583	2	12,050	71	14	15,764	2,701
Antimony	mg/kg dw	3	0.22	0.07	1	0.59	7	0.28	0.09	5	0.29	0.14	2	0.17	0.03	14	0.27	0.11
Arsenic	mg/kg dw	3	2.83	2.20	1	1.47	7	1.98	0.57	5	1.85	0.31	2	1.43	0.30	14	1.86	0.47
Barium	mg/kg dw	3	129	96	1	73.7	7	65.4	10.4	5	92.8	5.6	2	54.5	7.8	14	73.6	17.3
Beryllium	mg/kg dw	3	0.94	0.16	1	0.91	7	1.05	0.25	5	1.13	0.14	2	0.92	0.07	14	1.06	0.20
Bismuth	mg/kg dw	3	0.63	0.09	1	0.86	7	0.66	0.14	5	0.79	0.07	2	0.59	0.08	14	0.70	0.13
Boron	mg/kg dw	3	9.7	2.2	1	16.6	7	23.4	4.7	5	20.0	2.8	2	20.3	6.4	14	21.8	4.3
Cadmium	mg/kg dw	3	0.77	0.20	1	0.49	7	0.65	0.11	5	0.68	0.21	2	0.56	0.06	14	0.65	0.15
Calcium	mg/kg dw	3	3,443	1,087	1	3,570	7	4,864	502	5	4,440	178	2	4,410	283	14	4,648	428
Cesium	mg/kg dw	3	1.52	0.12	1	1.89	7	1.77	0.24	5	2.28	0.13	2	1.56	0.23	14	1.92	0.34
Chromium	mg/kg dw	3	38.0	3.3	1	34.1	7	32.6	2.6	5	38.0	1.7	2	28.6	1.5	14	34.0	4.0
Cobalt	mg/kg dw	3	34.9	27.4	1	11.3	7	14.1	4.3	5	11.4	1.7	2	22.5	21.0	14	14.3	7.5
Copper	mg/kg dw	3	109	3	1	107	7	104	14	5	106	4	2	114	16	14	106	11
Iron	mg/kg dw	3	61,633	61,509	1	23,100	7	32,771	12,231	5	26,320	5,283	2	53,700	56,710	14	33,457	20,197
Lead	mg/kg dw	3	5.20	1.28	1	4.97	7	5.64	0.76	5	5.38	0.59	2	5.13	0.64	14	5.47	0.67
Lithium	mg/kg dw	3	14.4	0.7	1	20.5	7	20.5	1.9	5	23.5	2.3	2	17.9	2.8	14	21.2	2.8
Magnesium	mg/kg dw	3	2,843	313	1	3,670	7	3,539	309	5	4,104	312	2	2,840	495	14	3,641	529
Manganese	mg/kg dw	3	2,451	3,861	1	242	7	227	55	5	244	57	2	161	25	14	223	57
Mercury	mg/kg dw	3	0.053	0.030	1	0.058	7	0.038	0.016	5	0.025	0	2	0.041	0.022	14	0.034	0.014
Molybdenum	mg/kg dw	3	11.0	4.2	1	7.74	7	11.1	4.0	5	9.6	1.7	2	11.4	7.1	14	10.6	3.6
Nickel	mg/kg dw	3	48.1	14.2	1	32.7	7	39.2	1.9	5	40.8	3.0	2	39.7	3.9	14	39.8	2.5
Phosphorus	mg/kg dw	3	1,657	741	1	1,510	7	1,071	203	5	1,110	181	2	1,035	304	14	1,080	192
Potassium	mg/kg dw	3	1,023	105	1	1,370	7	1,299	131	5	1,926	92	2	1,225	219	14	1,512	343
Rubidium	mg/kg dw	3	9.1	0.7	1	13.0	7	11.9	1.5	5	16.2	1.1	2	11.0	2.0	14	13.3	2.6
Selenium	mg/kg dw	3	1.96	0.23	1	1.73	7	1.61	0.26	5	1.71	0.08	2	1.89	0.45	14	1.69	0.24
Silver	mg/kg dw	3	0.18	0.07	1	0.22	7	0.20	0.16	5	0.12	0.04	2	0.16	0.08	14	0.17	0.12
Sodium	mg/kg dw	3	203	55	1	390	7	449	27	5	430	23	2	440	28	14	441	25
Strontium	mg/kg dw	3	31.8	8.0	1	39.8	7	73.0	11.1	5	57.9	5.5	2	56.1	0.1	14	65.2	11.5
Thallium	mg/kg dw	3	0.212	0.066	1	0.120	7	0.126	0.030	5	0.155	0.015	2	0.115	0.019	14	0.13	0.03
Tin	mg/kg dw	3	1.0	0.0	1	2.9	7	1.5	0.8	5	2.7	3.8	2	4.7	5.2	14	2.4	2.9
Titanium	mg/kg dw	3	202	22	1	244	7	216	54	5	244	40	2	229	25	14	228	46
Uranium	mg/kg dw	3	5.61	0.65	1	7.95	7	8.69	1.64	5	9.6	0.5	2	10.9	3.0	14	9.3	1.6
Vanadium	mg/kg dw	3	36.5	2.3	1	32.4	7	32.0	3.7	5	34.6	1.5	2	26.3	0.4	14	32.1	3.8
Zinc	mg/kg dw	3	156	21	1	115	7	148	29	5	143	20	2	122	16	14	143	25

Notes: Main basin is 14 near-field, mid-field, and far-field area stations.

n = sample size; SD = standard deviation of the mean; <= less than detection limit; N = nitrogen; P = phosphorus; S = sulphur; cm = centimetre;% dw = percent dry weight; mg/kg dw = milligrams per kilogram dry weight.

The most notable sediment concentration gradient for parameters among Snap Lake sampling areas was for available phosphate. Since 2008, the maximum available phosphate concentration has occurred at the diffuser station and then concentrations have decreased markedly in the other Snap Lake sampling areas and have been relatively low in Northeast Lake and Lake 13 sediments.

Within Snap Lake, the following spatial patterns were observed in 2012 with respect to occurrence of maximum mean parameter concentrations for the five lake areas:

- Parameters having their highest mean sediment concentrations at the diffuser or in the near-field area were TKN, available phosphate, available potassium, available sulphate, antimony, bismuth, boron, calcium, lead, mercury, silver, sodium, strontium, and titanium. The differences in concentration ranges between these and other lake areas were frequently not large, with the exception of available phosphate.
- The mid-field area had the highest mean concentrations of available nitrate, aluminum, beryllium, cesium, chromium, lithium, magnesium, potassium, rubidium, and titanium (also at diffuser).
- The far-field area had the highest mean concentrations of available ammonium, total nitrogen, copper, molybdenum, tin, and uranium. Mean total nitrogen and uranium concentrations have been elevated in the far-field area since 2004; 2010 was an exception, when one of the two far-field stations had high sand content and also low concentrations of a number of parameters (resulting in lower mean concentrations for the far-field area that year).
- The northwest arm had the highest mean concentrations of arsenic, barium, cadmium, chromium, cobalt, iron, manganese, nickel, phosphorus, selenium, thallium, vanadium, and zinc; exposure to treated effluent has been low in this part of Snap Lake.

When the 14 Snap Lake main basin stations were considered as a group, and compared to the diffuser and northwest arm areas, parameters having their highest mean sediment concentration in the main basin were available nitrate, available potassium, available sulphate, beryllium, boron, calcium, cesium, lead, lithium, potassium, rubidium, sodium, strontium, and uranium.

Of the 41 nutrients and metals analysed in 2012, mean concentrations of 22 parameters were higher in Northeast Lake or Lake 13 than in any areas of Snap Lake. Parameters having their maximum mean concentrations in Snap Lake, and parameters having their maximum mean concentrations in either Northeast Lake or Lake 13, are identified in Table 4-8.

Table 4-8 Occurrence of Maximum Mean Parameter Concentrations in Snap Lake, Northeast Lake, and Lake 13 in 2012

Parameters Having Maximum Area Mean Concentrations in Snap Lake	Parameters Having Maximum Mean Concentrations in Northeast Lake or Lake 13
Available nitrate, as N	Available ammonium, as N
Total Kjeldahl nitrogen (TKN)	Aluminum
Total nitrogen	Arsenic
Available phosphate, as P	Barium
Available potassium	Beryllium
Available sulphate, as S	Bismuth
Antimony	Cadmium
Boron	Cesium
Calcium	Chromium
Cobalt	Copper
Iron	Lead
Mercury	Lithium
Molybdenum	Magnesium
Phosphorus	Manganese
Selenium	Nickel
Silver	Potassium
Sodium	Rubidium
Strontium	Thallium
Tin	Titanium
	Uranium
	Vanadium
	Zinc

4.4.2.2 Sediment Depth Comparison (Top 5 cm and Top 2 cm)

The top 5 cm layer of sediment is currently sampled for sediment quality monitoring. However, 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.

Comparisons of sediment parameter concentrations in the top 5 cm versus the top 2 cm of sediment have now been performed on samples collected from a total of six Snap Lake stations in 2011 and 2012[9]: SNAP 14 and SNAP 15 (near-field) and SNAP20 (northwest arm) were sampled in 2011 (De Beers 2012); and SNP 02-20e (diffuser), SNAP 03 (near-field), and SNAP 17 (mid-field) were sampled in 2012.

For each parameter and sampling station, RPDs were calculated to provide a measure of the difference in concentrations between the two sampling depths (Table 4-9). RPDs are a measure

[9] The 2011 comparison used an Ekman grab to sample both sediment depths. The 2012 comparison used an Ekman grab to sample the top 5 cm layer and a Tech-Ops corer to sample the top 2 cm layer.

typically used to assess analytical precision through comparison of laboratory duplicate samples, with an RPD that is ≤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 in order to warrant modifying the study design to change the sediment sampling depth. Tin was excluded from this comparison because it was undetected in any samples from these six stations.

At the diffuser station (SNP 02-20e), 21 of the 40 nutrients or metals included for this comparison of sediment depths had RPDs that were >20% and the majority of RPDs were negative, which in this case meant that the sediment parameter concentration in the top 2 cm sample was higher than the concentration in the corresponding top 5 cm sample. One unexpected result was that the RPDs for available ammonium, available nitrate, available phosphate, and available potassium were large and positive, which meant that concentrations were lower in the top 2 cm sample; however, the analyses were repeated and the results were confirmed.

At the other five stations, at least 34 of the 40 nutrients or metals that were included for this sediment depth comparison had RPDs that were ≤20%, with the majority of those RPDs being <10%. The majority of RPDs were positive, meaning that the sediment parameter concentration in the top 2 cm sample was lower than the concentration in the corresponding top 5 cm sample.

Based on these results, the diffuser station was the only station where differences between the two sampling depths were large enough to be distinguishable from analytical variability and were indicative of a Mine-related effect. At the other stations, the differences in concentrations measured for the two sampling depths were small enough that they were not distinguishable from analytical variability associated with laboratory duplicate samples, and there was no clear pattern of concentrations being higher in shallower sediments, which would be expected if there was a Mine-related effect.

Table 4-9 Differences in Sediment Chemistry Between Top 5 cm and Top 2 cm Sediment Samples Collected in 2011 and 2012

Sampling Station	Units	SNP 02-20e (2012)			SNAP03 (2012)			SNAP 14 (2011)			SNAP15 (2011)			SNAP17 (2012)			SNAP20 (2011)		
Sediment Depth (cm)		Top 5 cm	Top 2 cm	RPD	Top 5 cm	Top 2 cm	RPD	Top 5 cm	Top 2 cm	RPD	Top 5 cm	Top 2 cm	RPD	Top 5 cm	Top 2 cm	RPD	Top 5 cm	Top 2 cm	RPD
Physical																			
Sand (>0.063 mm to <2.0 mm)	% dw	2.11	1.24	52%	0.18	0.18	0%	3.61	3.12	15%	2.89	3.26	-12%	6.96	6.46	7%	12.1	14.1	-15%
Silt (>0.004 mm to <0.063 mm)	% dw	88.1	89.5	-2%	88.9	91.2	-3%	47.9	47.4	1%	40.6	49.3	-19%	81.2	87.6	-8%	80.5	80.7	0%
Clay (<0.004 mm)	% dw	9.79	9.22	6%	10.9	8.62	23%	48.5	49.4	-2%	56.5	47.5	17%	11.9	5.92	67%	7.35	5.19	34%
Fines (Silt + Clay)	% dw	97.9	98.7	-1%	99.8	99.8	0%	96.4	96.8	0%	97.1	96.8	0%	93.1	93.5	0%	87.85	85.89	2%
Inorganic / Organic Carbon																			
Total Carbon	% dw	16.7	18.4	-10%	18.5	17.7	4%	16.7	17.7	-6%	16.2	15.9	2%	14.5	16.0	-10%	10.6	10.4	2%
Inorganic Carbon	% dw	0.12	<0.10	18%	<0.10	<0.10	0%	0.13	0.16	-21%	0.13	0.17	-27%	0.12	0.12	0%	0.2	0.2	0%
Total Organic Carbon	% dw	16.5	18.4	-11%	18.5	17.7	4%	16.6	17.5	-5%	16.1	15.7	3%	14.4	15.9	-10%	10.4	10.2	2%
Nutrients																			
Available Ammonium, as N	mg/kg dw	65	<25	89%	<19	<21	10%	88.1	85.2	3%	105	51.4	69%	23	<22	4%	68.6	49.6	32%
Total Kjeldahl Nitrogen (TKN)	% dw	1.33	1.51	-13%	1.32	1.40	-6%	1.27	1.35	-6%	1.22	1.18	3%	1.11	1.28	-14%	0.871	0.893	-2%
Total Nitrogen	% dw	1.28	1.54	-18%	1.41	1.39	1%	1.23	1.35	-9%	1.2	1.18	2%	1.08	1.27	-16%	0.861	0.851	1%
Available Nitrate, as N	mg/kg dw	44.0	10.4	124%	6.9	18.1	-90%	<6.0	<6.0	0%	<6.0	<6.0	0%	<4.0	9.5	-81%	<6.0	<6.0	0%
Available Phosphate, as P	mg/kg dw	426	14.1	187%	41.1	6.0	149%	18.6	17	9%	14.6	12	20%	38.4	20.1	63%	<4.0	<4.0	0%
Available Potassium	mg/kg dw	1320	269	132%	333	252	28%	120	171	-35%	145	138	5%	170	205	-19%	238	198	18%
Available Sulphate, as S	mg/kg dw	59.3	253	-124%	77.9	111	-35%	119	120	-1%	175	124	34%	134	160	-18%	208	197	5%
Metals																			
Aluminum	mg/kg dw	11500	11000	4%	12600	11500	9%	19600	18500	6%	18400	17700	4%	12700	11700	8%	12300	12200	1%
Antimony	mg/kg dw	0.12	0.38	-104%	0.23	0.27	-16%	0.13	0.15	-14%	0.2	0.19	5%	0.11	0.18	-48%	0.24	0.34	-34%
Arsenic	mg/kg dw	1.53	3.13	-69%	2.47	2.88	-15%	1.41	1.61	-13%	2.77	2.15	25%	1.82	2.43	-29%	6.91	6.97	-1%
Barium	mg/kg dw	68.9	76.8	-11%	54.7	49.3	10%	71.9	65.3	10%	83.1	77.4	7%	94.6	90.4	5%	520	737	-35%
Beryllium	mg/kg dw	0.69	0.50	32%	0.88	0.92	-4%	0.75	0.68	10%	0.95	0.8	17%	0.89	0.84	6%	0.38	0.39	-3%
Bismuth	mg/kg dw	0.68	0.77	-12%	0.77	0.72	7%	0.68	0.64	6%	0.78	0.73	7%	0.72	0.66	9%	0.47	0.45	4%
Boron	mg/kg dw	20.1	26.0	-26%	32.4	33.3	-3%	13	15.6	-18%	13.7	13.2	4%	22.0	24.1	-9%	4.6	4.3	7%
Cadmium	mg/kg dw	0.45	0.44	2%	0.55	0.48	14%	0.62	0.6	3%	0.8	0.75	6%	0.68	0.62	9%	0.59	0.57	3%
Calcium	mg/kg dw	3930	6490	-49%	5550	5950	-7%	4450	4880	-9%	5210	5400	-4%	4620	5390	-15%	2630	2930	-11%
Cesium	mg/kg dw	1.75	1.67	5%	1.73	1.65	5%	1.75	1.66	5%	1.92	1.88	2%	2.20	2.08	6%	1.26	1.21	4%
Chromium	mg/kg dw	30.8	38.7	-23%	29.1	28.3	3%	32.9	30.4	8%	30.7	30.5	1%	34.5	33.6	3%	25.9	25	4%
Cobalt	mg/kg dw	11.1	15.4	-32%	14.2	14.3	-1%	13.3	12.4	7%	17.6	15.2	15%	13.2	16.1	-20%	60	60.3	0%
Copper	mg/kg dw	106	94.8	11%	108	99.5	8%	109	102	7%	113	108	5%	99.9	90.0	10%	62.3	59.6	4%
Iron	mg/kg dw	17600	26300	-40%	34800	36500	-5%	33900	31500	7%	49300	44200	11%	19200	23500	-20%	220000	199000	10%
Lead	mg/kg dw	5.33	10.4	-64%	6.45	7.27	-12%	5.39	6.64	-21%	7.97	6.78	16%	5.60	6.34	-12%	8.37	8.51	-2%
Lithium	mg/kg dw	20.4	22.9	-12%	20.2	18.6	8%	23.8	23	3%	21.1	20.8	1%	24.7	24.1	2%	10.9	10.4	5%
Magnesium	mg/kg dw	2920	5790	-66%	2920	2940	-1%	4080	3970	3%	3470	3560	-3%	3620	3510	3%	1760	1750	1%
Manganese	mg/kg dw	246	373	-41%	259	490	-62%	249	223	11%	552	721	-27%	273	652	-82%	27800	41800	-40%
Mercury	mg/kg dw	0.062	0.101	-48%	0.056	0.065	-15%	<0.050	<0.050	0%	0.051	<0.050	0%	<0.050	<0.050	0%	0.085	0.091	-7%
Molybdenum	mg/kg dw	9.18	13.5	-38%	14.8	15.8	-7%	11.9	10.9	9%	13.8	12.9	7%	9.78	11.4	-15%	16.5	16.2	2%
Nickel	mg/kg dw	33.1	53.9	-48%	43.9	45.3	-3%	37.3	36	4%	43.7	40.7	7%	44.1	47.8	-8%	43.6	41.5	5%
Phosphorus	mg/kg dw	1510	1620	-7%	1280	1250	2%	840	800	5%	1050	1000	5%	1020	1050	-3%	1350	1350	0%
Potassium	mg/kg dw	1380	1470	-6%	1290	1260	2%	1740	1650	5%	1530	1470	4%	2190	2190	0%	940	1000	-6%
Rubidium	mg/kg dw	13.3	13.0	2%	12.3	11.6	6%	12.6	12.1	4%	12.5	12.1	3%	18.2	17.2	6%	7.2	7.3	-1%
Selenium (by ICPMS)	mg/kg dw	1.77	2.03	-14%	1.85	1.85	0%	0.87	0.79	10%	1.18	1.12	5%	1.48	1.65	-11%	1.07	1.06	1%
Silver	mg/kg dw	0.23	0.39	-52%	0.21	0.23	-9%	0.2	0.22	-10%	0.2	<0.20	0%	<0.20	<0.20	0%	<0.20	<0.20	0%
Sodium	mg/kg dw	440	810	-59%	560	620	-10%	460	490	-6%	470	430	9%	460	520	-12%	120	130	-8%
Strontium	mg/kg dw	44.9	110	-84%	82.0	88.3	-7%	51.8	64.3	-22%	72.5	71.3	2%	62.4	73.9	-17%	32.7	37.4	-13%
Thallium	mg/kg dw	0.135	0.127	6%	0.105	0.082	25%	0.138	0.121	13%	0.129	0.112	14%	0.178	0.169	5%	0.133	0.138	-4%
Titanium	mg/kg dw	245	248	-1%	192	165	15%	209	189	10%	180	175	3%	269	277	-3%	177	179	-1%
Uranium	mg/kg dw	8.27	8.80	-6%	9.89	9.48	4%	10.3	9.54	8%	9.89	9.61	3%	9.16	8.54	7%	3.76	3.59	5%
Vanadium	mg/kg dw	29.9	29.6	1%	30.1	28.2	7%	33.5	30.8	8%	33.4	31.7	5%	33.1	31.5	5%	25.9	25.1	3%
Zinc	mg/kg dw	110	102	8%	154	149	3%	147	136	8%	146	132	10%	159	140	13%	97	95.6	1%

RPD = relative percent difference; N = nitrogen: <= less than detection limit; - = not analyzed/not applicable;% dw = percent dry weight; mg/kg dw = milligrams per kilogram based on dry weight; ICPMS = inductively coupled plasma mass spectrometry; cm = centimetre; mm = millimetre, P = phosphorus; TKN = total Kjeldahl nitrogen.

4.4.3 Comparison to Sediment Quality Guidelines

Of the parameters analyzed in Snap Lake and Northeast Lake sediment samples in 2012, Canadian SQGs were available for seven metals (Table 4-4). Concentrations of a number of those metals were above SQGs in Snap Lake sediments in 2012, as was observed in previous years, and also in Northeast Lake and Lake 13 sediments (Table 4-10). Because the number of stations sampled varied in 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 in Snap Lake have only occasionally been above the ISQG since 2004 (Table 4-10), at stations located in the diffuser, near-field, or northwest arm areas. Since 2007, exceedance of the arsenic ISQG has only occurred at SNAP20 in the northwest arm, and only in 2007, 2009, and 2011. Bottom conductivity at SNAP20 has been close to the minimum in all years, and a contribution from the Mine discharge can therefore be eliminated.

Concentrations of cadmium, chromium, and copper in Snap Lake sediments were above the ISQG in at least 40% of the 2004 baseline sediment samples (Table 4-10). Therefore, it appears that these metals are naturally elevated in Snap Lake sediments. Consistent with the 2004 results, concentrations of cadmium and copper have been above ISQGs between 2005 and 2012, in generally similar proportions as observed in 2004, indicating natural sources. Chromium concentrations in Snap Lake were above the ISQG in all years except 2009; the frequency of occurrence was lower in top 5 cm samples than in bulk samples in previous years, again indicating natural sources.

Lead concentrations in Snap Lake have been below the ISQG since 2004, with the exception of anomalously high results in 2005 for two diffuser and near-field stations. Those lead concentrations of 161 mg/kg dw 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 Snap Lake sediments were above the ISQG in all the 2004 baseline sediment samples (Table 4-10), and above the PEL at mid-field station SNAP17 in 2004. As with cadmium, chromium, and copper, it appears that zinc is naturally elevated in Snap Lake sediments. Zinc concentrations have been above the ISQG between 2005 and 2012, although generally less frequently than in 2004.

Table 4-10 Guideline Exceedances for Metals in Snap Lake, 2004 to 2012, Northeast Lake, 2008 to 2012, and Lake 13, 2012

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	-	-	-	-	-	-	-
Northeast Lake	2008 Top 5 cm	ISQG	5	-	5/5 (100%)	4/5 (80%)	5/5 (100%)	-	-	5/5 (100%)
		PEL	5	-	-	-	-	-	-	-
	2009 Top 5 cm	ISQG	5	-	5/5 (100%)	5/5 (100%)	5/5 (100%)	-	-	5/5 (100%)
		PEL	5	-	-	-	-	-	-	-
	2010 Top 5 cm	ISQG	5	-	5/5 (100%)	5/5 (100%)	5/5 (100%)	-	-	5/5 (100%)
		PEL	5	-	-	-	-	-	-	-
	2011 Top 5 cm	ISQG	5	-	4/5 (80%)	5/5 (100%)	5/5 (100%)	-	-	5/5 (100%)
		PEL	5	-	-	-	-	-	-	-
	2012 Top 5 cm	ISQG	5	-	5/5 (100%)	5/5 (100%)	5/5 (100%)	-	-	5/5 (100%)
		PEL	5	-	-	-	-	-	-	-
Lake 13	2012 Top 5 cm	ISQG	5	4/5 (80%)	1/5 (20%)	5/5 (100%)	5/5 (100%)	-	-	-
		ISQG	5	2/5 (40%)	-	-	-	-	-	-

Notes: Percentage in parentheses indicates the percentage of stations where the sediment concentration was above the relevant guideline.

A dash (-) indicates that no stations had sediment concentrations exceeding the guideline; if the number of stations for which data were available was less than the number of stations sampled, that number is shown in parentheses after the dash.

n = number of stations sampled. 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); ISQG= Interim Sediment Quality Guideline; PEL= Probable Effect Level; cm= centimetre;%=percent.

In Northeast Lake sediments, concentrations of cadmium, chromium, copper, and zinc were above their ISQGs at all five stations in 2012. This was consistent with the results obtained since monitoring of sediment quality in Northeast Lake began in 2008, and provides further evidence of naturally elevated concentrations of these metals in sediments in the area surrounding the Mine.

In Lake 13 sediments, concentrations of lead, mercury, and zinc were below their respective ISQGs at all five stations in 2012, and cadmium was above its ISQG at only one station. Concentrations of chromium and copper were above their ISQGs at all five stations; this was consistent with observations for Snap Lake and Northeast Lake and suggests that concentrations are naturally elevated in sediments in the area surrounding the Mine. Arsenic concentrations were higher in Lake 13 sediments than either Snap Lake or Northeast Lake, ranging from 5.0 to 37.2 mg/kg; concentrations were above the ISQG at four stations and above the PEL at two stations. Arsenic concentrations measured in three Lake 13 sediment samples in July 2005 ranged from 4.0 to 6.2 mg/kg (Golder 2005). However, the EAR reported maximum arsenic sediment concentrations in the Lockhart River watershed of 49.0 mg/kg in 1993/1994 and 55.3 mg/kg in August 1999 (De Beers 2002). Thus, the Lake 13 arsenic sediment concentrations are well within natural variability.

4.4.4 Spatial Patterns in Snap Lake Sediment Quality in 2012

Conductivity Gradient

Of the 41 sediment quality parameters analyzed using linear regression, concentrations of available phosphate, antimony, bismuth, chromium, mercury, nickel, and phosphorus varied significantly with bottom conductivity (Table 4-11). The regression equations for chromium and nickel had negative slopes, indicating a decrease in concentration with increasing exposure to the treated effluent, which was inconsistent with an effect on sediment quality associated with the treated effluent discharge. The regression equations for available phosphate, antimony, bismuth, mercury, and phosphorus had positive slopes, which implied increasing parameter concentrations in sediments with increasing concentration of the treated effluent at the lake bottom, as indicated by the shaded *P*-values in Table 4-11.

The 2006 bottom conductivity data, used for the linear regressions because of conductivity's strong gradient, were plotted against sediment concentrations of available phosphate, antimony, bismuth, mercury, and phosphorus (Figure 4-5). The corresponding 2004 bulk sediment data were also plotted on each graph for matching stations (except available phosphate, which was not analysed in 2004), to investigate the possibility of pre-existing spatial trends in sediment quality that may mimic Mine-related effects.

A comparison of the slopes of the regression lines for the 2012 and 2004 data for antimony, bismuth, mercury, and phosphorus revealed that effects on sediment quality from exposure to treated effluent were unlikely because the slopes for these four parameters were not statistically significantly different ($P<0.10$). Available phosphate could not be evaluated as there were no corresponding 2004 data for this parameter.

Table 4-11 Results of Linear Regressions for Sediment Quality Data, and 2004 vs. 2012 Slope Comparisons

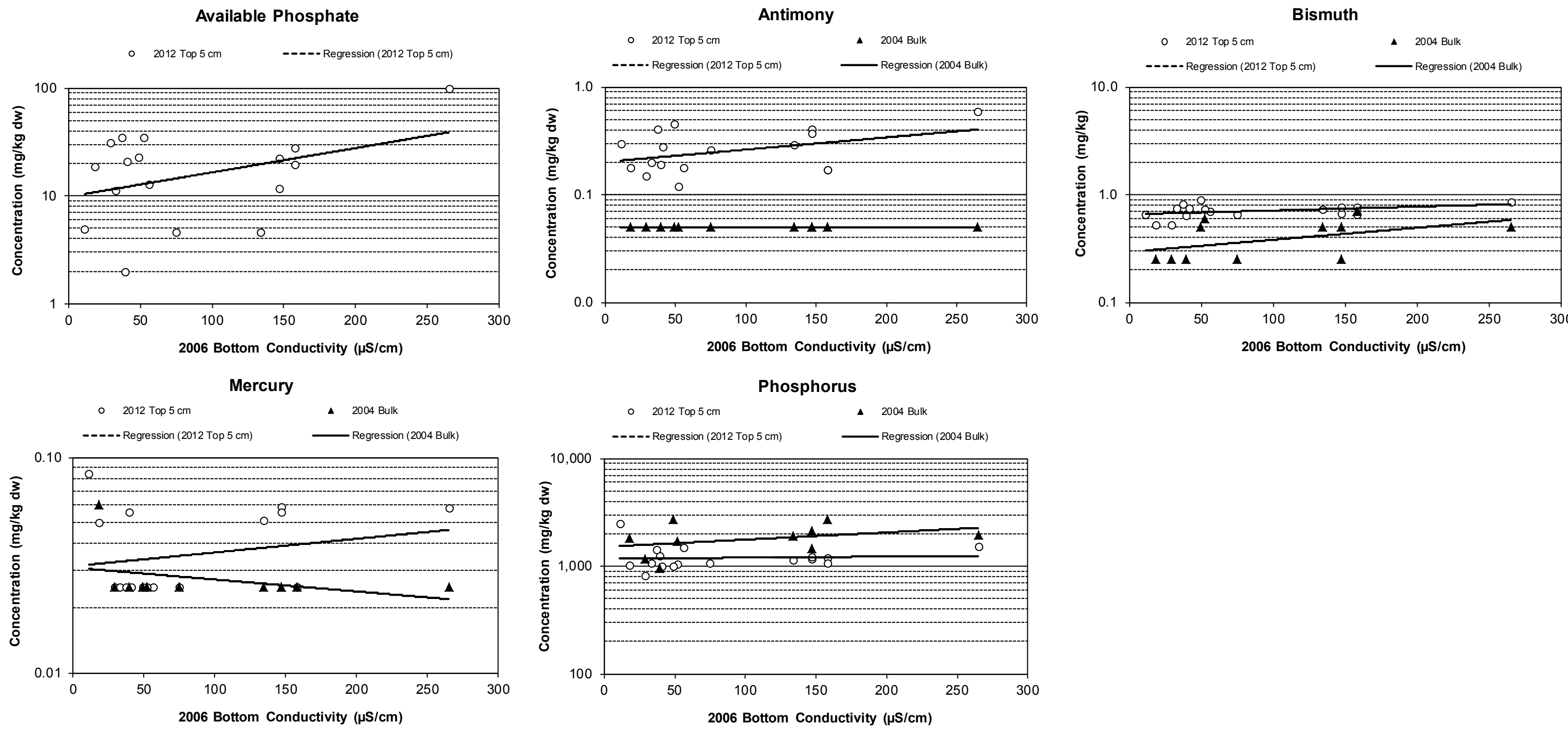
Parameter	Units	Linear Regression Results					2004 Slope vs. 2012 Slope (*P*-value)
		n	Slope	Intercept	R^2	*P*-value	
Nutrients							
Available ammonium, as N	mg/kg dw	17/16	-0.073/-0.045	31.57/26.41	0.078/0.051	0.276/0.399	-
Available nitrate, as N	mg/kg dw	17	0.025	8.81	0.032	0.490	-
Total Kjeldahl nitrogen (TKN)	% dw	17	0.001	1.25	0.109	0.196	-
Total nitrogen	% dw	17/16	0.0001	1.36/1.43	0.0001/0.056	0.984/0.376	-
Available phosphate, as P	mg/kg dw	17	0.184	6.87	0.334	**0.015**	Not tested
Available potassium	mg/kg dw	17/16	0.082/0.029	110.6/111.1	0.042/0.008	0.431/0.743	-
Available sulphate, as S	mg/kg dw	17/16	0.216/-0.022	130.7/128.4	0.019/0.001	0.5990.926	-
Total Metals							
Aluminum	mg/kg dw	17	-1.44	16,540	0.003	0.837	-
Antimony	mg/kg dw	17	0.001	0.201	0.236	**0.048**	0.169
Arsenic	mg/kg dw	17/16	-0.003/0.001	2.268/1.773	0.038/0.011	0.452/0.705	-
Barium	mg/kg dw	17/16	-0.210/-0.059	102.3/79.92	0.119/0.069	0.176/0.326	-
Beryllium	mg/kg dw	17	0.0001	1.060	0.001	0.916	-
Bismuth	mg/kg dw	17/16	0.001/0.001	0.664/0.642	0.153/0.227	0.121/**0.036**	0.235-
Boron	mg/kg dw	17	0.026	17.21	0.087	0.250	-
Cadmium	mg/kg dw	17	-0.001	0.720	0.121	0.171	-
Calcium	mg/kg dw	17/16	1.418/-0.786	4,207/4,532	0.020/0.014	0.590/0.657	-
Cesium	mg/kg dw	17	0.0001	1.873	0.001	0.902	-
Chromium	mg/kg dw	17/16	-0.015/-0.021	38.39/37.38	0.090/0.226	0.242/**0.063**	-
Cobalt	mg/kg dw	17/16	-0.059/-0.013	23.0/16.2	0.084/0.013	0.260/0.674	-
Copper	mg/kg dw	17/16	0.004/0.015	108.3/106.4	0.002/0.048	0.850/0.416	-
Iron	mg/kg dw	17/16	-110.2/-18.29	48,367/34,800	0.066/0.005	0.319/0.802	-
Lead	mg/kg dw	17/16	0.0001/-0.001	5.339/5.356	0.002/0.018	0.880/0.620	-
Lithium	mg/kg dw	17	0.010	19.1	0.038	0.450	-
Magnesium	mg/kg dw	17	0.577	3,421	0.005	0.782	-
Manganese	mg/kg dw	17/16	-6.237/0.079	1,151/219.2	0.073/0.011	0.295/0.702	-
Mercury	mg/kg dw	17/16	0.0001	0.035/0.027	0.032/0.211	0.496/**0.074**	0.209
Molybdenum	mg/kg dw	17	-0.004	11.23	0.008	0.737	-
Nickel	mg/kg dw	17/16	-0.048/-0.026	44.91/41.69	0.250/0.387	**0.041/0.010**	-
Phosphorus	mg/kg dw	17/16	-0.125/1.205	1,244/1,048	0.001/0.199	0.928/**0.084**	0.412
Potassium	mg/kg dw	17	-0.691	1,504	0.019	0.599	-
Rubidium	mg/kg dw	17	0.001	12.73	0.001	0.923	-
Selenium	mg/kg dw	17	-0.001	1.819	0.042	0.429	-
Silver	mg/kg dw	17/16	0.0001	0.161/0.134	0.012/0.063	0.681/0.350	-
Sodium	mg/kg dw	17/16	0.462/0.233	356.5/390.3	0.111/0.045	0.192/0.429	-
Strontium	mg/kg dw	17	0.071	50.7	0.092	0.237	-
Thallium	mg/kg dw	17/16	0.0001	0.174/0.156	0.197/0.151	**0.074**/0.137	-
Tin	mg/kg dw	17	-0.003	2.541	0.008	0.725	-
Titanium	mg/kg dw	17/16	-0.090/-0.052	225.7/217.2	0.036/0.022	0.464/0.586	-
Uranium	mg/kg dw	17/16	0.002/0.005	8.663/8.157	0.007/0.051	0.744/0.402	-
Vanadium	mg/kg dw	17/16	-0.007/-0.013	33.95/34.46	0.023/0.107	0.565/0.216	-
Zinc	mg/kg dw	17/16	-0.084/-0.075	151.6/146.9	0.055/0.078	0.363/0.295	-

Note: *P*-values indicating significant ($P<0.1$) linear regressions (positive or negative) are highlighted in **bold**, and *P*-values with significant positive slopes are shaded grey.

P-values indicating significant differences ($P<0.1$) between 2004 and 2012 slopes are highlighted in **bold**.

n = number of samples for each parameter used in the regression analysis; $\mathbf{R}^2$ = coefficient of determination; N = nitrogen; P = phosphorus; S = sulphur;% dw = percent dry weight; mg/kg dw = milligrams per kilogram dry weight; – = not applicable because slope of significant regression line was negative.

Figure 4-5 Scatter-Plots for Parameters Having Significant Regression with Positive Slope



cm = centimetre; mg/kg dw = milligrams per kilogram dry weight; µS/cm= microSiemens/centimetre.

Among-Area Spatial Comparisons

Mean sediment parameter concentrations in the main basin of Snap Lake were compared to mean concentrations in the reference lakes; 2012 was the first year these statistical analyses were performed. For those parameters where the mean main basin concentration was greater than the mean concentration in either Northeast Lake or Lake 13, statistical tests were used to compare these areas. Results of these statistical analyses are summarized in Table 4-12.

Statistically significant differences ($P<0.10$) among areas overall were identified for available nitrate, available sulphate, TKN, total nitrogen, aluminum, antimony, beryllium, boron, calcium, copper, selenium, sodium, strontium, and zinc. Results of the planned comparisons ($P<0.03$) between areas were:

- Sediment quality parameters with significantly higher mean concentrations in the main basin as compared to the pooled reference means for Northeast Lake and Lake 13 were total nitrogen, antimony, boron, copper, selenium, sodium, and strontium.
- There were statistically significant differences between Northeast Lake and Lake 13 for available nitrate, available sulphate, TKN, total nitrogen, antimony, beryllium, boron, calcium, copper, selenium, sodium, strontium, and zinc.
- Sediment quality parameters with significantly higher mean concentrations in the main basin as compared to Northeast Lake were TKN, calcium, sodium, and strontium.

4.4.5 Temporal Trends in Snap Lake Sediment Quality

When evaluating temporal trends, it was assumed that an effect due to discharge of treated effluent would be easiest to detect by comparing years with clear stepwise increases in treated effluent exposure. As indicated by the bottom conductivity data in Figure 4-4, such increases occurred in the near-field area from 2004 to 2005 and from 2006 to 2007, and in the mid-field and far-field areas from 2006 to 2007. Exposure to treated effluent could also result in a gradual build-up of treated effluent-associated parameters in bottom sediments, which would be most prominent in the diffuser and near-field areas, where cumulative exposure to the treated effluent has been highest since the beginning of treated effluent discharge.

For most sediment quality parameters, clear temporal trends in concentrations were not observed in the near-field and mid-field areas of Snap Lake between 2004 and 2012 (Appendix 4B, Figure 4B-2). Both increases and decreases in mean concentrations occurred during this period. Mean concentrations of total nitrogen, arsenic, barium, cadmium, cesium, chromium, magnesium, nickel, potassium, rubidium, thallium, titanium, and zinc decreased in both the near-field and mid-field areas.

Table 4-12 Results of Statistical Tests Comparing Sediment Quality in Snap Lake Main Basin With Reference Lakes, 2012

Parameter (mg/kg dw)	Overall ANOVA Test Result [a] (*P*-value)	Planned Comparisons [b]			Magnitude of Difference		
		Main Basin vs Pooled NEL and LK13 (*P*-value)	NEL vs LK13 (*P*-value)	Main Basin vs NEL (*P*-value)	Main Basin from Pooled NEL and LK13 (%)	NEL from LK13 (%)	Main Basin from NEL (%)
Available nitrate [c]	**0.058**	0.400	**0.023**	0.050	260	55	196
Available phosphate	0.535	0.273	0.689	0.370	10	-40	47
Available potassium [c]	0.832	0.528	0.542	0.746	-2	-11	4
Available sulphate	**<0.0001**	0.123	**<0.0001**	0.225	147	292	55
Total Kjeldahl nitrogen (TKN)	**<0.0001**	0.841	**<0.0001**	**0.002**	58	44	34
Total nitrogen [c]	**0.001**	**0.019**	**<0.0001**	0.032	44	75	13
Aluminum	**0.053**	**0.016**	0.590	0.031	-6	23	-15
Antimony	**0.004**	**0.008**	**0.007**	0.161	11	138	-21
Beryllium	**0.001**	**<0.0001**	**0.023**	**0.008**	-2	62	-21
Boron [c]	**0.003**	**0.006**	**<0.0001**	0.310	64	194	10
Calcium [c]	**<0.0001**	0.868	**<0.0001**	**<0.0001**	58	50	31
Copper	**<0.0001**	**<0.0001**	**<0.0001**	**<0.0001**	7	108	-21
Mercury	0.832	0.549	0.864	0.610	-5	16	-12
Molybdenum	0.313	0.630	0.133	0.770	18	29	5
Phosphorus	0.639	0.416	0.517	0.626	-1	-10	5
Selenium	**<0.0001**	**0.008**	**<0.0001**	0.930	24	65	-1
Silver [c]	0.301	0.843	0.213	0.663	37	44	16
Sodium [c]	**<0.0001**	**<0.0001**	**<0.0001**	**<0.0001**	177	-11	194
Strontium [c]	**<0.0001**	**0.004**	**<0.0001**	**<0.0001**	148	14	132
Tin	0.423	0.915	0.266	0.505	86	56	52
Zinc	**<0.0001**	**<0.0001**	**0.015**	**0.001**	-5	70	-25

Note: *P*-values representing statistically significant differences are **bold**.

(a) Analysis of Variance (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 $P<0.03$ after a Dunn-Ŝidák correction of an original *P*-value of 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.

NEL = Northeast Lake; LK13 = Lake 13; Main Basin = main basin of Snap Lake; *P*-value = probability;% = percent; mg/kg dw = milligrams per kilogram dry weight.

Selenium concentrations were less than 0.1 mg/kg dw in 2004 and 2005, but increased approximately 10-fold in 2006, and have fluctuated subsequently. ALS confirmed that there was no change in the methodology used by EnviroTest Laboratories in 2005 or by ALS in 2006 for the selenium analyses. The selenium methodology changed from hydride to ICP-MS in 2009, but a comparison of methods undertaken in 2010 (De Beers 2011) indicated that this change should not affect comparability of data among years from 2006 onward.

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 2012, although these trends were not all statistically significant. Baseline 2004 data were not available for the diffuser area. The parameters for which statistically significant temporal trends were identified (either increasing or decreasing) in Snap Lake or Northeast Lake, are summarized in Table 4-13 along with their significance level (*P*). Statistically significant positive trends were identified for 17 parameters in one or more areas of Snap Lake, or in Northeast Lake. Of the seven parameters that had statistically significant positive trends at the diffuser area, only available potassium, bismuth, and selenium also had statistically significant positive trends in other sampling areas within Snap Lake.

Mean concentrations of available potassium, available sulphate, antimony, bismuth, boron, calcium, iron, lithium, mercury, molybdenum, selenium, silver, sodium, strontium, and uranium had significantly positive temporal trends in one or more areas within Snap Lake. Of those parameters, the only ones to show significantly positive temporal trends in the Snap Lake main basin were available potassium, available sulphate, bismuth, selenium, sodium, and strontium. The overlap and variability in mean area concentrations for the parameters showing significant positive trends in the Snap Lake main basin from 2004 to 2012 are illustrated in Figure 4-6; mean concentrations for Northeast Lake and Lake 13 are also included for comparison. Bismuth had a statistically significant positive relationship with bottom conductivity in 2012, but it was not significantly different from the 2004 data. Mean sodium and strontium concentrations in the main basin were significantly higher than in Northeast Lake in 2012.

The Snap Lake sampling area means for 2005 to 2012 were above the normal ranges, estimated as the baseline whole-lake mean ±2SD (Table 4-14), for a number of parameters in one or more years. This included occurrences of mean concentrations for the northwest arm being above baseline ranges. Because exposure to the treated effluent has been relatively low in this area, this finding suggests that the baseline range was not fully represented by the 2004 data set.

Mean concentrations of available phosphate, antimony, bismuth, mercury, selenium, silver, sodium, strontium, and tin were above their respective normal ranges in the diffuser and/or near-field areas of Snap Lake in 2012. Mean concentrations of antimony, bismuth, cobalt, iron, manganese, selenium, sodium, strontium, and tin were also above baseline ranges in one or more other areas of Snap Lake in 2012.

Table 4-13 Summary of Statistically Significant Temporal Trends in Snap Lake Sediments for 2004 to 2012, and Northeast Lake Sediments for 2008 to 2012

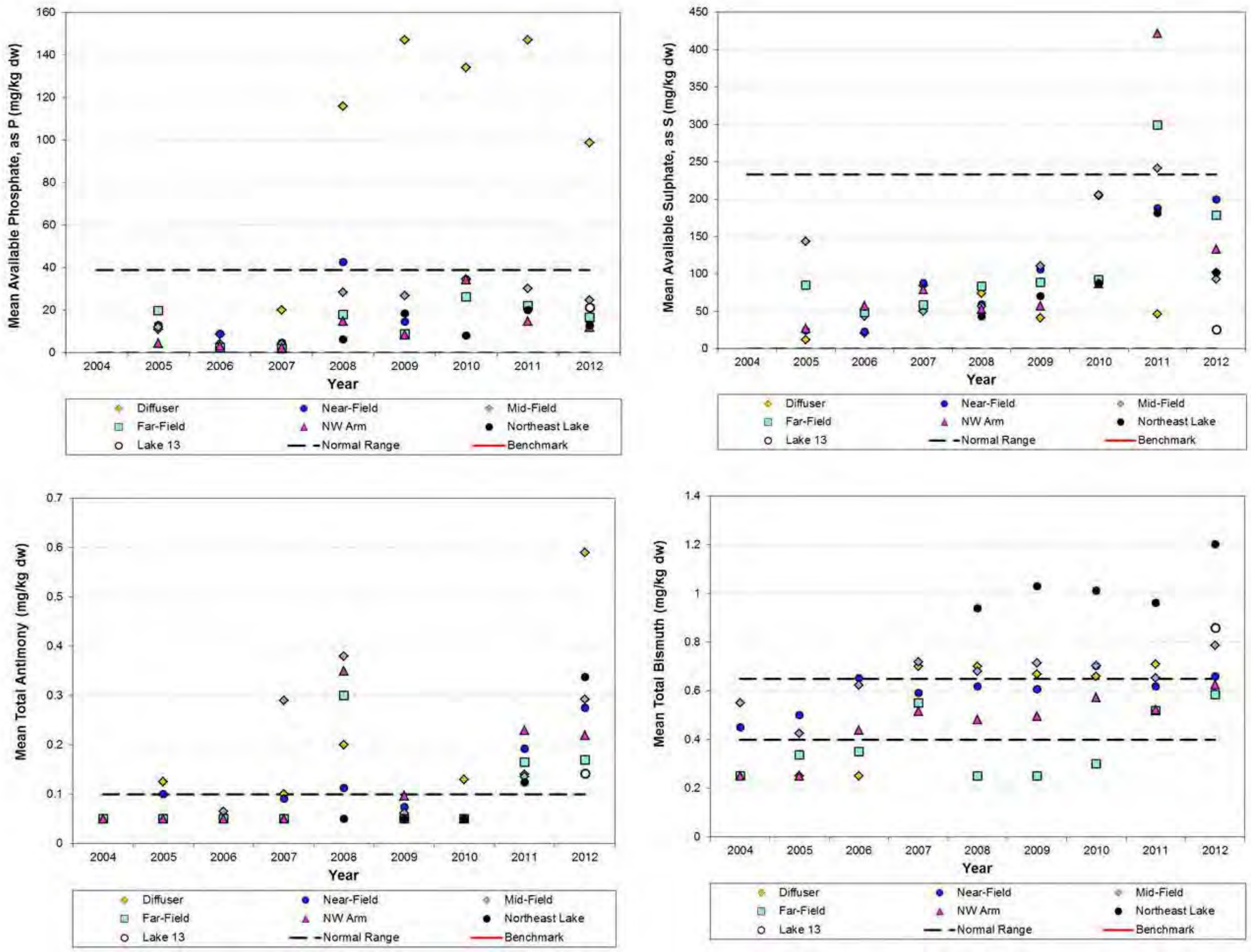
Parameter	Direction and Significance Level (*P*) of Statistically Significant Temporal Trends						
	Snap Lake Sampling Area						Northeast Lake
	Northwest Arm	Diffuser	Near-Field	Mid-field	Far-field	Main Basin	
Nutrients	-	-	-	-	-	-	-
Available Ammonium, as N	-	-	-	-	-	-	
Available Nitrate, as N	DEC (*P*<0.05)	-	-	-	-	-	INC (*P*<0.10)
Total Kjeldahl Nitrogen (TKN)	-	-	-	-	-	-	-
Total Nitrogen	-	-	-	-	-	-	-
Available Phosphate, as P	-	-	-	-	-	-	-
Available Potassium	-	INC (*P*<0.01)	INC (*P*<0.05)	INC (*P*<0.10)		INC (*P*<0.05)	
Available Sulphate, as S	INC (*P*<0.05)	-	INC (*P*<0.01)	-	INC (*P*<0.05)	INC (*P*<0.05)	INC (*P*<0.10)
Metals	-	-	-	-	-	-	-
Aluminum	-	-	-	-	-	-	-
Antimony	-	-	INC (*P*<0.10)	-	-	-	-
Arsenic	-	DEC (*P*<0.05)	DEC (*P*<0.01)	DEC (*P*<0.05)	-	DEC (*P*<0.05)	-
Barium	DEC (*P*<0.10)	DEC (*P*<0.01)	DEC (*P*<0.001)	DEC (*P*<0.01)	DEC (*P*<0.05)	DEC (*P*<0.001)	-
Beryllium	-	-	-	-	-	-	-
Bismuth	INC (*P*<0.01)	INC (*P*<0.10)	INC (*P*<0.10)	INC (*P*<0.10)	-	INC (*P*<0.05)	-
Boron	-	-	-	INC (*P*<0.05)	-	-	-
Cadmium	-	-	-	-	-	DEC (*P*<0.10)	-
Calcium	-	-	INC (*P*<0.10)	-	-	-	-
Cesium	-	DEC (*P*<0.10)	DEC (*P*<0.05)	DEC (*P*<0.05)	-	DEC (*P*<0.05)	-
Chromium	-	-	-	-	-	-	-
Cobalt	-	-	-	-	-	-	-
Copper	-	-	-	-	-	-	INC (*P*<0.10)
Iron	-	INC (*P*<0.10)	-	-	-	-	-
Lead	-	-	-	-	-	-	-
Lithium	-	-	-	-	INC (*P*<0.10)	-	-
Magnesium	-	-	-	-	-	-	-

Table 4-13 Summary of Statistically Significant Temporal Trends in Snap Lake Sediments for 2004 to 2012, and Northeast Lake Sediments for 2008 to 2012

Parameter	Direction and Significance Level (*P*) of Statistically Significant Temporal Trends						
	Snap Lake Sampling Area						Northeast Lake
	Northwest Arm	Diffuser	Near-Field	Mid-field	Far-field	Main Basin	
Manganese	-	-	-	-	-	-	-
Mercury	-	INC (*P*<0.10)	-	-	-	-	-
Molybdenum	-	INC (*P*<0.10)	-	-	-	-	INC (*P*<0.05)
Nickel	-	-	-	-	-	-	-
Phosphorus	-	-	-	-	-	-	-
Potassium	-	DEC (*P*<0.10)	-	-	-	DEC (*P*<0.01)	-
Rubidium	-	-	DEC (*P*<0.05)	-	-	DEC (*P*<0.05)	-
Selenium	INC (*P*<0.05)	INC (*P*<0.05)	INC (*P*<0.05)	INC (*P*<0.05)	INC (*P*<0.05)	INC (*P*<0.05)	-
Silver	INC (*P*<0.05)	-	INC (*P*<0.01)	-	-	-	-
Sodium	-	-	INC (*P*<0.05)	INC (*P*<0.05)	INC (*P*<0.10)	INC (*P*<0.01)	-
Strontium	-	-	INC (*P*<0.001)	INC (*P*<0.01)	INC (*P*<0.05)	INC (*P*<0.001)	INC (*P*<0.10)
Thallium	-	-	DEC (*P*<0.05)	DEC (*P*<0.05)	DEC (*P*<0.05)	DEC (*P*<0.10)	-
Titanium	DEC (*P*<0.05)	DEC (*P*<0.05)	DEC (*P*<0.01)	-	-	DEC (*P*<0.01)	DEC (*P*<0.10)
Uranium	-	INC (*P*<0.10)	-	-	-	-	-
Vanadium	-	-	-	-	-	-	-
Zinc	-	-	DEC (*P*<0.05)	DEC (*P*<0.05)	DEC (*P*<0.05)	DEC (*P*<0.05)	-

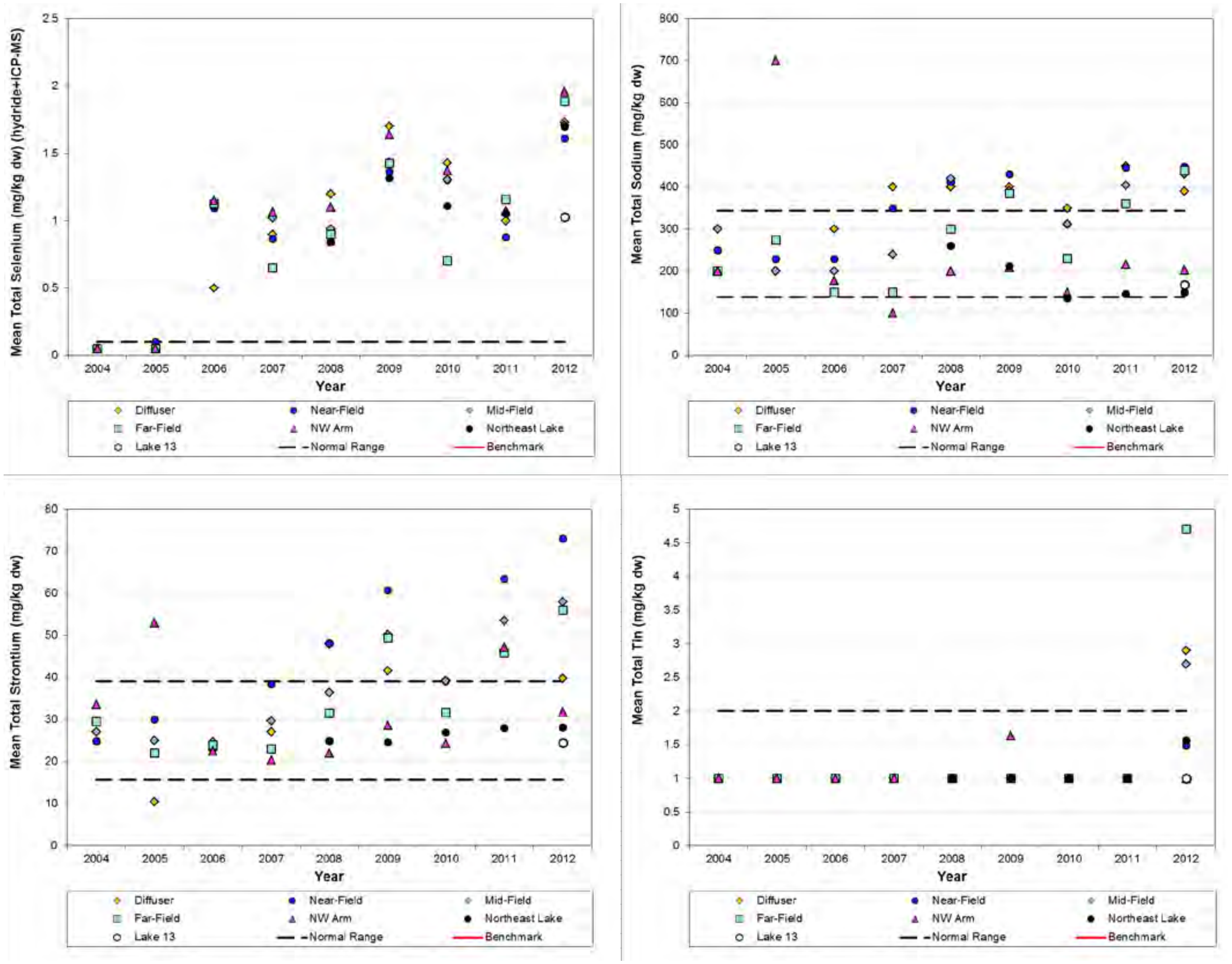
"-" = dashes indicate no statistically significant (*P*<0.10) temporal trend present; DEC = decreasing; INC = increasing; N = Nitrogen; P = Phosphorus; S = Sulphate.

Figure 4-6 Plots of Temporal Trends in Snap Lake Sampling Areas for Selected Parameters, 2004 to 2012



mg/kg dw = milligrams per kilogram dry weight; P = Phosphorus; S = Sulphate.

Figure 4-6 Plots of Temporal Trends in Snap Lake Sampling Areas for Selected Parameters, 2004 to 2012



mg/kg dw = milligrams per kilogram dry weight.

Table 4-14 Comparison of 2005 to 2012 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							
		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
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, MF, FF	-	-	-	NWA, D, MF, FF	-
Available Potassium	mg/kg dw	2005	9	91.8	105.0	41.0	159.0	32.0	27.7 - 156	-	-	-	-	-	-	NWA	-
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, NF	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, MF, FF	
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)	mg/kg dw	2004	12	<0.10	<0.10	<0.10	<0.10	0.00	0.10 - 0.10	D	-	MF	NWA, MF, FF		D	NWA, D, NF, MF, FF	NWA, D, NF, MF, FF, MB
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, MF	D, MF	D, MF	MF	D	D, NF, MF, 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	-	-	-	-	-	-	-
Cesium (mg/kg) [(b)]	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 , FF	NWA, FF
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, FF	NWA, FF
Lead (mg/kg)	mg/kg dw	2004	12	5.5	5.2	3.5	9.7	1.6	2.4 - 8.6	D, NF	-	-	-	-	-	-	-
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	-	-	-	-	-	-	-
Manganese (mg/kg)	mg/kg dw	2004	12	287	279	146	434	96	96 - 478		NWA	NWA	NWA	NWA	NWA	NWA	NWA
Mercury (mg/kg)	mg/kg dw	2004	12	0.05	<0.05	<0.05	0.06	0.00	0.05 - 0.06	MF	NWA	NWA			NWA, D	NWA, 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) [(a,c)]	mg/kg dw	2004	12	<0.10	<0.10	<0.10	<0.10	0.00	0.10 - 0.10	-	NWA, D, NF, MF, FF	NWA, D, NF, MF, FF	NWA, D, NF, MF, FF	NWA, D, NF, MF, FF	NWA, D, NF, MF, FF	NWA, D, NF, MF, FF	NWA, D, NF, MF, FF, MB
Silver [(a)]	mg/kg dw	2004	12	<0.20	<0.20	<0.20	<0.20	0.00	0.20 - 0.20	-	-	-	-	-	D	D	D
Sodium (mg/kg)	mg/kg dw	2004	12	242	200	200	300	51	139 - 345	NWA	-	D, NF	D, NF, MF	D, NF, MF, FF	D, NF	D, NF, MF, FF	D, NF, MF, FF, MB
Strontium (mg/kg)	mg/kg dw	2004	12	27.4	26.0	21.0	42.0	5.8	15.7 - 39.1	NWA	-	-	-	D, NF, MF, FF	D, NF, MF	NWA, D, NF, MF, FF	D, NF, MF, FF, MB
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) [(a)]	mg/kg dw	2004	12	<2.00	<2.00	<2.00	<2.00	0.00	2.00 - 2.00	-	-	-	-	-	-	-	D, MF, FF, 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) 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.
(b) 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.
(c) Selenium was analyzed by the hydride method from 2004 to 2008, and by inductively coupled plasma mass spectrometry (ICP-MS) from 2009 to 2012.
"-" = dashes indicate that lake area mean concentrations were within normal ranges; n = number of samples; SD = Standard deviation; % = percent;% 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; NF = Near-field Area; MF = Mid-field Area; FF = Far-field Area; MB = main basin of Snap Lake; - = not applicable.

Mean concentrations of antimony, bismuth, selenium, sodium, strontium, and tin in the main basin were outside their normal ranges, and are included in Figure 4-6. Antimony, silver, and tin were not detected in baseline sediment samples and therefore their normal ranges are equal to their respective DLs. Antimony and silver have been detected more frequently in recent years, and tin was detected in multiple samples for the first time in 2012.

Overall, evaluation of temporal trends in sediment quality did not provide clear evidence of an effect on Snap Lake sediments in areas exposed to treated effluent from the Mine, although mean concentrations of available potassium, available sulphate, bismuth, iron, mercury, molybdenum, selenium, sodium, strontium, and uranium demonstrated statistically significant positive trends in mean concentrations at the diffuser and/or in the main basin of Snap Lake.

The magnitude and pattern of some of these 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, 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. Concentrations of sediment quality parameters were generally similar in the diffuser area as compared to other areas of Snap Lake in 2012, although the marked gradient in available phosphate concentrations observed since 2008 in the diffuser area continued in 2012. Increasing trends in concentrations of certain parameters were balanced by decreasing concentrations of other parameters, with most parameters exhibiting no clear trends.

4.4.6 Summary

Sediment quality monitoring in Snap Lake from 2004 to 2012 documented variability in concentrations of most parameters, both among stations and years. The 2012 concentration ranges were within a factor of five for most parameters. Snap Lake sediments are generally characterized as being primarily fine-grained material with elevated TOC concentrations, although grain size composition has not been consistent for all years. This inconsistency was previously addressed by De Beers (2006, 2007) and identified as being due to differences in the methods used by the analytical laboratory in 2005 and 2006 for grain size analysis. Sediments from Northeast Lake exhibited similar characteristics to Snap Lake sediments. Although Lake 13, which was evaluated as a provisional second reference lake in 2012, had sediments that were also primarily fine-grained materials, the TOC concentrations were lower than in Snap Lake and Northeast Lake.

Concentrations of a number of metals with available SQGs were above those SQGs in Snap Lake in 2012, which was also observed in previous AEMP years and in 2004 under baseline conditions. Concentrations of lead and mercury were consistently below ISQGs. Concentrations of cadmium, chromium, copper, and zinc are naturally elevated in Snap Lake and Northeast Lake sediments and frequently above ISQGs, although chromium concentrations in Snap Lake sediments were below their ISQG in 2009. Arsenic concentrations were occasionally above their ISQG in Snap Lake after 2004, but did not appear to be related to the Mine. Cadmium, chromium, and copper concentrations in Lake 13 sediments were above ISQGs; notably, arsenic concentrations were higher in Lake 13 than in Snap Lake or Northeast Lake, and were above their ISQG at four stations and above their PEL at two stations.

In 2012, seven sediment parameters had concentrations that varied significantly with Snap Lake bottom conductivity. Chromium and nickel were negatively related to bottom conductivity; however, available phosphate, antimony, bismuth, mercury, and phosphorus were positively related to bottom conductivity. Additional analyses to investigate the possibility of pre-existing spatial trends in sediment quality that mimic Mine-related effects indicated that a spatially related effect on sediment quality from exposure to treated effluent was unlikely for four of the five parameters; this relationship could not be tested for available phosphate due to lack of baseline data.

Temporal trends were evaluated in each sampling area, with particular attention to the near-field and mid-field areas, where clear stepwise increases in treated effluent exposure occurred between 2004 and 2007. Temporal trends were also assessed for the main basin of Snap Lake. In the diffuser area, concentrations of most parameters were higher in 2012 than in 2005, but this difference reflected low concentrations in 2005 and 2006, rather than elevated concentrations in 2012. Clear temporal trends between 2004 and 2012 were not observed in concentrations of most sediment quality parameters in the near-field and mid-field areas. Increasing trends in concentrations of certain parameters were balanced by decreasing concentrations of others, while most parameters exhibited no clear trends. Temporal trends possibly indicative of a Mine-related effect were observed, specifically increasing mean concentrations of available potassium, available sulphate, bismuth, iron, mercury, molybdenum, selenium, sodium, strontium, and uranium at the diffuser and/or the main basin of Snap Lake over time. Mean concentrations of available nitrate, available sulphate, copper, molybdenum, and strontium also increased significantly in Northeast Lake between 2008 and 2012.

Snap Lake sampling area means for 2005 to 2012 were compared with the estimated normal ranges in sediment quality parameter concentrations. The northwest arm means for a number of parameters analyzed from 2005 to 2012 were above their respective normal ranges. Because this area has received the lowest exposure to treated effluent, this result suggests that the normal range was not fully represented by the 2004 baseline data set. In 2012, mean concentrations of available phosphate, antimony, bismuth, mercury, selenium, silver, sodium, strontium, and tin were above their respective normal ranges in the diffuser area. Concentrations of antimony, bismuth, selenium, sodium, strontium, and tin were also above their respective normal ranges in the main basin of Snap Lake.

Overall, evaluation of spatial and temporal patterns in sediment quality did not provide clear evidence of an effect on Snap Lake sediments in areas exposed to treated effluent from the Mine. Ongoing sediment quality monitoring under the AEMP is expected to provide a more reliable indication of any potential effects on sediment quality in Snap Lake as the number of years of available data increases. If potential effects to sediment quality have occurred to date, they have been subtle and not clearly different than natural variability.

4.5 CONCLUSIONS

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 cadmium, chromium, copper, and zinc were above their respective ISQGs at one or more Snap Lake stations and at all five Northeast Lake stations in 2012. Similar results were reported in previous years for both lakes, and in the 2004 baseline survey for Snap Lake, indicating that these metals occur naturally at concentrations above their respective ISQGs. Concentrations of arsenic, lead, and mercury were below their respective ISQGs in both lakes in 2012.

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?

Spatial patterns occurred within Snap Lake but spatial distributions of sediment quality parameters were unlikely to be related to the Mine, based on comparison to baseline conditions. Available phosphate, antimony, bismuth, mercury, and phosphorus concentrations varied significantly with exposure to treated effluent based on comparison to bottom conductivity; however, comparison to baseline conditions did not identify a significant relationship with increasing exposure to treated effluent. The concentration of available phosphate in the diffuser area underwent a large increase between 2007 and 2008, and then increased gradually through 2012; concentrations in other areas of Snap Lake decreased markedly with increasing distance from the diffuser but remained higher than in 2007.

4.5.3 Are Concentrations of Sediment Quality Parameters Increasing over Time?

Most sediment quality parameters are not increasing over time. Although statistically significant positive trends were observed for available potassium, available sulphate, antimony, bismuth, boron, calcium, iron, lithium, mercury, molybdenum, selenium, silver, sodium, strontium, and uranium in one or more areas of Snap Lake, the magnitude of these temporal changes was generally either not large or mean concentrations have remained consistent since 2007 following an initial increase. Only available potassium, available sulphate, bismuth, selenium, sodium, and strontium had significant positive trends in the Snap Lake main basin. Antimony, mercury, and silver concentrations have been undetected in most samples in previous years; these parameters have been detected with greater frequency since 2010 but at concentrations close to their respective DLs. Tin was detected in sediments from several areas within Snap Lake for the first time in 2012.

4.6 RECOMMENDATIONS

Recommendations for modifications to the sediment quality component of the Snap Lake AEMP are identified below.

- Continue to use Northeast Lake as a reference lake to assess long-term regional trends.
- Repeat sediment quality sampling in Lake 13 in 2013 to determine whether the elevated and variable concentrations of arsenic, barium, and manganese observed in 2012 are representative of actual conditions in Lake 13.

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5 PLANKTON

5.1 INTRODUCTION

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 taxonomic 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. Microcystin is a hepatotoxin (i.e., liver toxin) that can be produced by several common genera of cyanobacteria (e.g., *Aphanizomenon*, *Microcystis*, *Anabaena*, *Oscillatoria*). However, the presence of these cyanobacterial species does not necessarily mean microcystin will be produced (Aboal and Puig 2005). Microcystin-LR, which contains the amino acids lysine [L] and arginine [R], is the most toxic and commonly measured parameter to assess microcystin production (WHO 2003, 2011). 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 Aquatic Effects Monitoring Program (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 continued as required by the Water License MV2001L2-0002 (MVLWB 2004), and the renewed Water Licence MV2011L2-0004 (MVLWB 2012).

Light penetration into a waterbody is a function of the absorption and scattering of light in the water. Dissolved or colloidal materials, inorganic clastic (i.e., erosional) materials and phytoplankton will scatter and absorb light, reducing the water's transparency. Secchi depth measurements are measured in the AEMP to determine light penetration. However, Secchi depth measurements are subjective and depend on the environmental conditions at the time of measurement; thus they provide relative, not absolute information on light penetration. For the purposes of the AEMP, the Secchi depth is both used as a surrogate for phytoplankton growth, as in many waterbodies. Secchi depth is inversely related to phytoplankton biomass (Dodds and Whiles 2010), and as a measure of the aesthetic characteristics of light (Kirk 1994).

5.1.1.2 Zooplankton

The term “zooplankton” refers to microscopic animals and includes Rotifera (rotifers) and crustaceans, specifically Cladocera (cladocerans or water fleas), 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.

Zooplankton community composition and biomass can also be affected by changes in the environment. Increased phytoplankton biomass can translate into increased food availability for zooplankton and, ultimately, fish. However, fish and predatory zooplankton species presence 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 algae. *Daphnia* spp. are large zooplankton capable of intense grazing and may be responsible for triggering seasonal blooms of larger colonial phytoplankton species, such as species of cyanobacteria and chlorophytes (Lampert et al. 1986).

The grazing rate of copepods is lower than cladocerans; as a result, copepods do not have as great of an effect on the phytoplankton community structure (Wetzel 2001).

5.1.1.3 Picoplankton

Picoplankton are the smallest size category of plankton ranging between 0.2 to 2 µm in size. They include two major groups: free living heterotrophic bacteria, and small autotrophic phytoplankton, with the most ubiquitous being pico-cyanobacteria. Picoplankton play an important role in total carbon production and biomass in oligotrophic lakes (Sinistro et al. 2007). They also provide a rich food source for zooplankton, which ultimately translates into food resources for fish.

Picoplankton are important contributors to the “microbial loop”, which is a model of pathways for nutrient and carbon cycling by microbial components in the pelagic community (e.g., bacteria, picoplankton, micro-ciliates, as well as autotrophic, mixotrophic, and heterotrophic nano-flagellates). They are, therefore, also affected by changes in nutrient concentrations in the system. Picoplankton are sensitive indicators of nutrient enrichment, owing to their small size and simple cellular structure, which results in a high growth rate and increased efficiency in nutrient uptake (Schallenberg and Burns 2001). Growth rates of autotrophic picoplankton in ultra-oligotrophic and meso-oligotrophic lakes have been shown to be inhibited by additions of phosphorus (P) and nitrogen (N; Stockner and Shortreed 1994; Schallenberg and Burns 2001). Colonial forms of autotrophic picoplankton are commonly found in productive oligotrophic and mesotrophic lakes, which suggests that these colonial forms are preferred in times of nutrient depletion and may provide refuge from grazing (Stockner 1991; Schallenberg and Burns 2001). As such, picoplankton can be an effective early warning indicator of environmental change (Munawar and Weisse 1989; Stockner 1991).

5.1.1.4 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 understood by examining the distribution, supply, and composition of these nutrients in Snap Lake. Although toxicity is also a possibility from other substances introduced into Snap Lake by the Mine, the primary effect of the Mine on plankton communities is nutrient enrichment (De Beers 2012a; Section 13), which is the focus of this section of the 2012 AEMP report.

High 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 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).

The approximate ratios that N and P are generally found in are reflected in the Redfield Ratio (molar ratio of 106 carbon [C]:16N:1P; Wetzel 2001). An N:P molar ratio greater than 22 indicates P-limitation, an nitrogen to phosphorus (N:P) ratio of less than 13 indicates N-limitation, and ratios between 13 and 22 indicate nutrient sufficiency for optimal algal growth (Hillebrand and Sommer 1998). Cellular nutrient concentrations of natural phytoplankton communities can reveal the type and extent of nutrient limitation and requirements, while molar ratios of dissolved nutrient concentrations reflect water column availability. If total nutrients are used in the computation of the molar ratios, the predictive value of nutrient ratios becomes more complex, as both cellular and dissolved forms are represented. Cellular nutrients are affected by light energy available for photosynthesis, and the dissolved forms are affected by the biogeochemical processes (i.e., uptake, sedimentation, and loading) occurring in the system (Wetzel 2001). However, despite their complexity, these molar ratios are useful as predictive measures for phytoplankton development.

In addition to total nitrogen (TN) and total phosphorus (TP), soluble reactive Si concentrations measured during the AEMP water quality program were also included in this assessment as changes in Si concentration may result in changes in phytoplankton community composition. Silica is a nutrient required for diatom growth. 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 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).

5.1.1.5 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. These two components of the plankton community are the main monitoring tools used in the plankton program. Picoplankton abundance can be used as an even earlier indicator of nutrient changes. The usefulness of picoplankton as a secondary monitoring component to assess nutrient enrichment has been investigated as a Special Study since 2008 (De Beers 2009, 2010, 2011, 2012a). Shared methods are used for the collection of picoplankton and phytoplankton, and data analyses and results of the picoplankton, phytoplankton, and zooplankton data are interconnected; therefore, the Picoplankton Special Study is presented herein as part of the plankton component, rather than as a separate Special Study. The inherent variability within the plankton community poses a challenge and also limits their usefulness as a monitoring tool. Plankton density, biomass, and species 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 Objectives

The principal objective of the plankton (phytoplankton and zooplankton) monitoring component of the AEMP for the Snap Lake Mine (Mine) is to meet the following specific Water Licence MV2011L2-0004 (Part G, Schedule 6, Item 1) (MVLWB 2012) 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. Concentration 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.*

The following five questions were addressed relative to the above Water Licence objectives for phytoplankton and zooplankton:

1 What are the current concentrations of chlorophyll *a* and *c*, and what do these concentrations indicate about the trophic status of Snap Lake and Northeast Lake?

2 What is the current status, in terms of abundance, biomass, and composition, of the phytoplankton community in Snap Lake and Northeast Lake, and do these results suggest Mine-related nutrient enrichment or toxicological impairment?

3 What is the current status, in terms of abundance, biomass, and composition, of the zooplankton community in Snap Lake and Northeast Lake, and do these results suggest signs of Mine-related nutrient enrichment or toxicological impairment?

4 How do observed changes compare to applicable predictions in the Environmental Assessment Report (EAR)?

5 How does the plankton community in reference Lake 13 compare to Snap Lake and Northeast Lake? Is reference Lake 13 a suitable reference lake for plankton?

The following additional two key questions were specific to picoplankton monitoring:

6 What is the current status, in terms of abundance, of the picoplankton community in Snap Lake, Northeast Lake, and Lake 13, and do these results provide any evidence of Mine-related nutrient enrichment or toxicological impairment?

7 How do the observed changes in the picoplankton community compare to changes observed in the phytoplankton community?

In addition to addressing the seven key questions listed above, plankton community characteristics were compared to predictions in the EAR (De Beers 2002). The EAR predicted:

- a lake-wide gradual increase in TP from 4 to 12 µg/L to 13 to 23 µg/L, with TP remaining in the lower to mid-range accepted for mesotrophic lakes;
- a gradual increase in chlorophyll *a* (a measure of the quantity of algae) 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 total dissolved solids (TDS) concentration, which would lead to an increase in calcium concentrations (to 110 milligrams per litre [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 METHODS

5.2.1 Field Survey

5.2.1.1 Changes to the Original Study Design

Changes were made to the number and location of sampling stations for the plankton program in 2012. These changes were implemented to provide as much overlap as possible in sampling stations between the water quality and plankton components. With better station integration, the supporting limnological data collected by the water quality component can be used by the plankton component, which reduces overall redundancy between the programs. Results of the water quality component (Section 3) indicate treated effluent mixing has occurred throughout the main basin of Snap Lake and plankton station-to-station variability is low (De Beers 2012a), supporting the changes in sampling stations noted below.

In Snap Lake, the number of stations increased by two in the northwest arm and remained the same in the main basin. Three plankton stations were relocated to the closest water quality stations as follows: SNAP13 was grouped with SNAP02-2e; SNAP31 was grouped with SNAP29; and, SNAP11 was grouped with SNAP11A. Two new stations, SNAP02A and SNAP20B, were added to the northwest arm of Snap Lake, while SNAP12 was removed from the plankton program.

Based on information collected during a review of potential reference lakes in 2005 (Golder 2005), Lake 13 was determined to be the second most similar lake to Snap Lake, following Northeast Lake, on the basis of size, shape, and physical characteristics. Inclusion of a second reference lake in the AEMP study design provides a better basis upon which to determine whether changes in Snap Lake are natural or Mine-related. In August 2012, a single station (LK13-01) was sampled in Lake 13 to assess whether it is an acceptable reference lake for plankton sampling.

5.2.1.2 Sampling Locations

In 2012, plankton sampling occurred at ten monitoring stations within Snap Lake, including five stations in the main basin of Snap Lake (SNAP02-20e, SNAP03, SNAP06, SNAP11A, and SNAP08) and five stations in the northwest arm (SNAP29, SNAP02A, SNAP01, SNAP20B, and SNAP30; Figure 5-1). Plankton monitoring stations in Snap Lake were numbered according to the common comprehensive numbering system that was applied to all components of the AEMP (De Beers 2005). Gaps in the station numbering occurred because some sampling stations were part of a different monitoring program (e.g., benthic invertebrate stations), which did not fully overlap with the plankton program.

In Northeast Lake, the plankton program was completed at five monitoring stations (NEL01, NEL02, NEL03, NEL04, and NEL05; Figure 5-2). These sampling stations are consistent with the water quality component. Plankton sampling also occurred at one station in Lake 13 (LK13-01; Figure 5-3).

5.2.1.3 Timing of Sampling

To accurately assess seasonal variability of the Snap Lake and Northeast Lake plankton communities, sampling occurred monthly during the open-water season, between July and September. To determine whether plankton parameters in Lake 13 were similar to Snap Lake or Northeast Lake, samples were collected during a single sampling event on August 8, 2012. A summary of the sampling events completed in Snap Lake, Northeast Lake, and Lake 13 are presented in Table 5-1.

5.2.1.4 Issues with Sample Collection in 2012

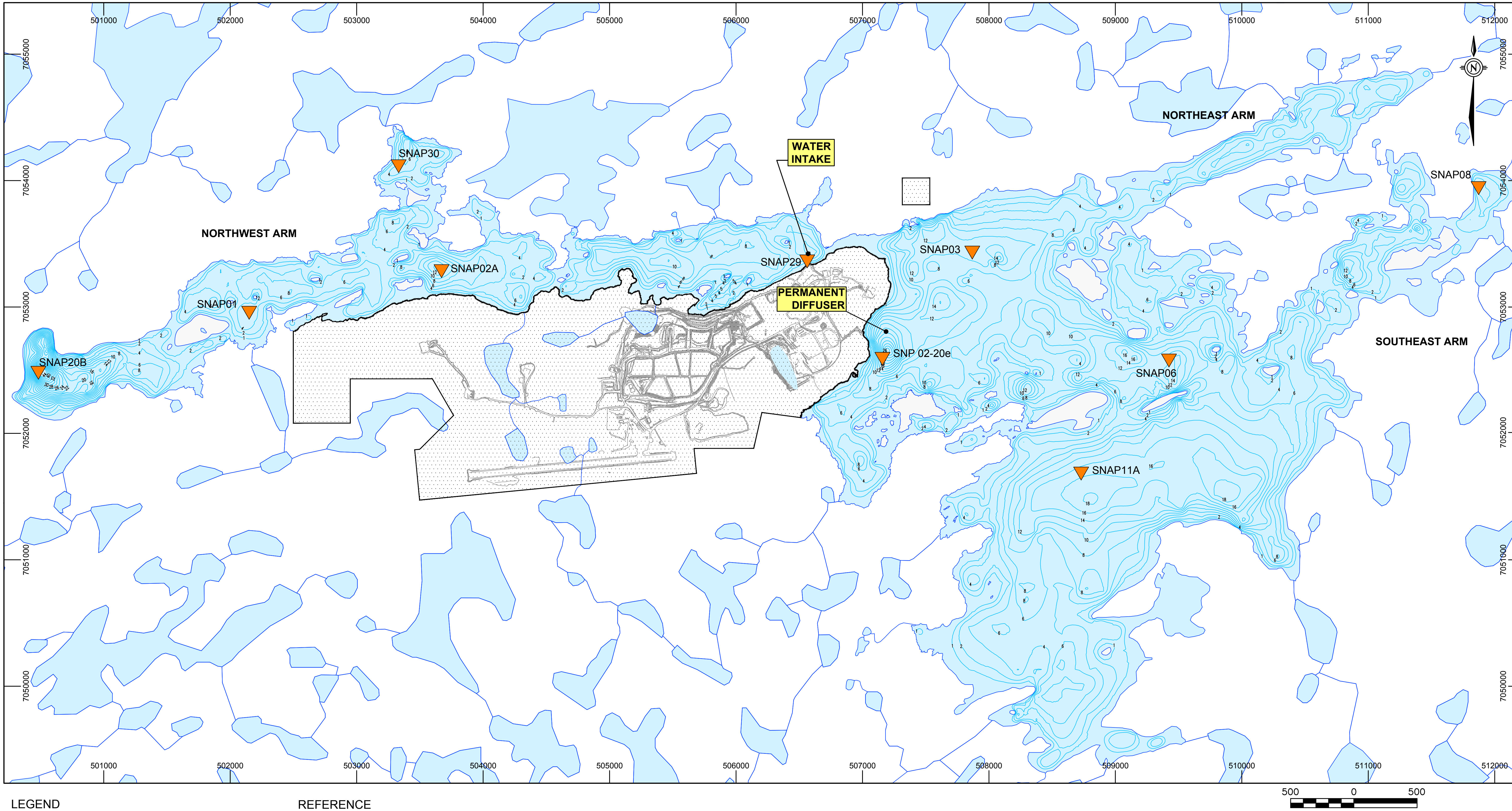
A number of sample-related issues occurred during the 2012 plankton program. These issues were related to sample contamination, confusion in sampling requirements at a single plankton station, and confusion in the submission of samples to the analytical laboratories on one occasion.

In July 2012, laboratory contamination of a bottle of deionized reference water, which is used to create trip and equipment blanks for the water quality component (Section 3), compromised samples collected at SNAP08. The samples were collected and sent to the laboratory before the contamination issue was identified. The contamination issue only affected one plankton chlorophyll *a* and *c* sample at SNAP08, but it affected water quality samples (Section 3). Because the nutrient samples for plankton and water quality were being collected at the same time due to the Nutrient Special Study (Section 12.4) requirements, all samples that were affected by contamination in the water quality component were flagged, the samples were discarded, and laboratory analyses were not performed. A number of the stations were resampled; however, during the resampling process the field crew overlooked resampling SNAP08 for depth-integrated chlorophyll *a* and *c*.

Water profile data were not collected for SNAP01 in August. It is exclusively a plankton sampling station and it does not coincide with a water quality sampling station. Because SNAP01 is not a water quality sampling station, the field crew overlooked the need to collect water profile data for the plankton component. In addition, at SNAP 11A and SNAP08, dissolved oxygen (DO) was recorded as percent (%) DO rather than in mg/L. Therefore, DO values for those stations were removed from the data set.

Nutrient samples were collected for NEL03, NEL04, and NEL05 in September in combination with the Nutrient Special Study (Section 12.4). A sample submission error caused confusion among the three laboratories that were part of the Nutrient Special Study. This error would have resulted in uncertainties in the data. Therefore, the depth-integrated TN/TP samples were discarded.

Although there were some field collection issues, on balance the 2012 field program for the plankton component was completed successfully. Streamlining between the plankton, water quality, and nutrient special study components was maximized. Although detailed specific work instructions were provided to all field crew members, the complexity of these three components was substantial and may have overwhelmed the field crews. Future monitoring programs will require a pre-field meeting to discuss the upcoming program as well as to identify any issues from the previous sampling program and identify preventive measures.



LEGEND

- PLANKTON MONITORING STATION
- SNAP LAKE MINE FOOTPRINT
- WATERBODY
- DEPTH CONTOUR (m)

REFERENCE

DIGITAL MAP FROM MACKAY LAKE, NORTHWEST TERRITORIES, PRODUCED BY DEPARTMENT OF ENERGY, MINES AND RESOURCES. MAP 75M, ORIGINAL SCALE 1:250,000, NAD 83 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. MAP MOVED TO NAD83 COORDINATE SYSTEM.

PROJECT

DE BEERS
GROUP OF COMPANIES

TITLE

PLANKTON MONITORING STATIONS IN SNAP LAKE, 2012

Golder Associates

PROJECT	13-1349-0001.4100	FILE No.	13134900014100A5-1
DESIGN	TD	2013-03-13	SCALE AS SHOWN REV. 0
CADD	JEF	2013-03-30	
CHECK	LMB	2013-04-19	FIGURE 5-1
REVIEW	PMC	2013-04-19	

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LEGEND

PLANKTON AEMP MONITORING STATION

DEPTH CONTOUR (m)

WATERBODY

REFERENCES

DIGITIZED FROM NTS TOPOGRAPHIC MAP 75 M/10 © 1985 HER MAJESTY THE QUEEN IN RIGHT OF CANADA. DEPARTMENT OF ENERGY, MINES AND RESOURCES. PROJECTION : TRANSVERSE MERCATOR, DATUM : NAD27, COORDINATE SYSTEM : UTM ZONE 12.

REFERENCE 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.

500 0 500

SCALE METRES

PROJECT

DE BEERS
GROUP OF COMPANIES

TITLE

PLANKTON MONITORING STATIONS IN NORTHEAST LAKE, 2012

Golder Associates

PROJECT	13-1349-0001.4100	FILE No.	13134900014100A5-2
DESIGN	TD	2013-03-13	SCALE AS SHOWN REV. 0
CADD	JEF	2013-03-30	
CHECK	LMB	2013-04-19	
REVIEW	PMC	2013-04-19	

FIGURE 5-2

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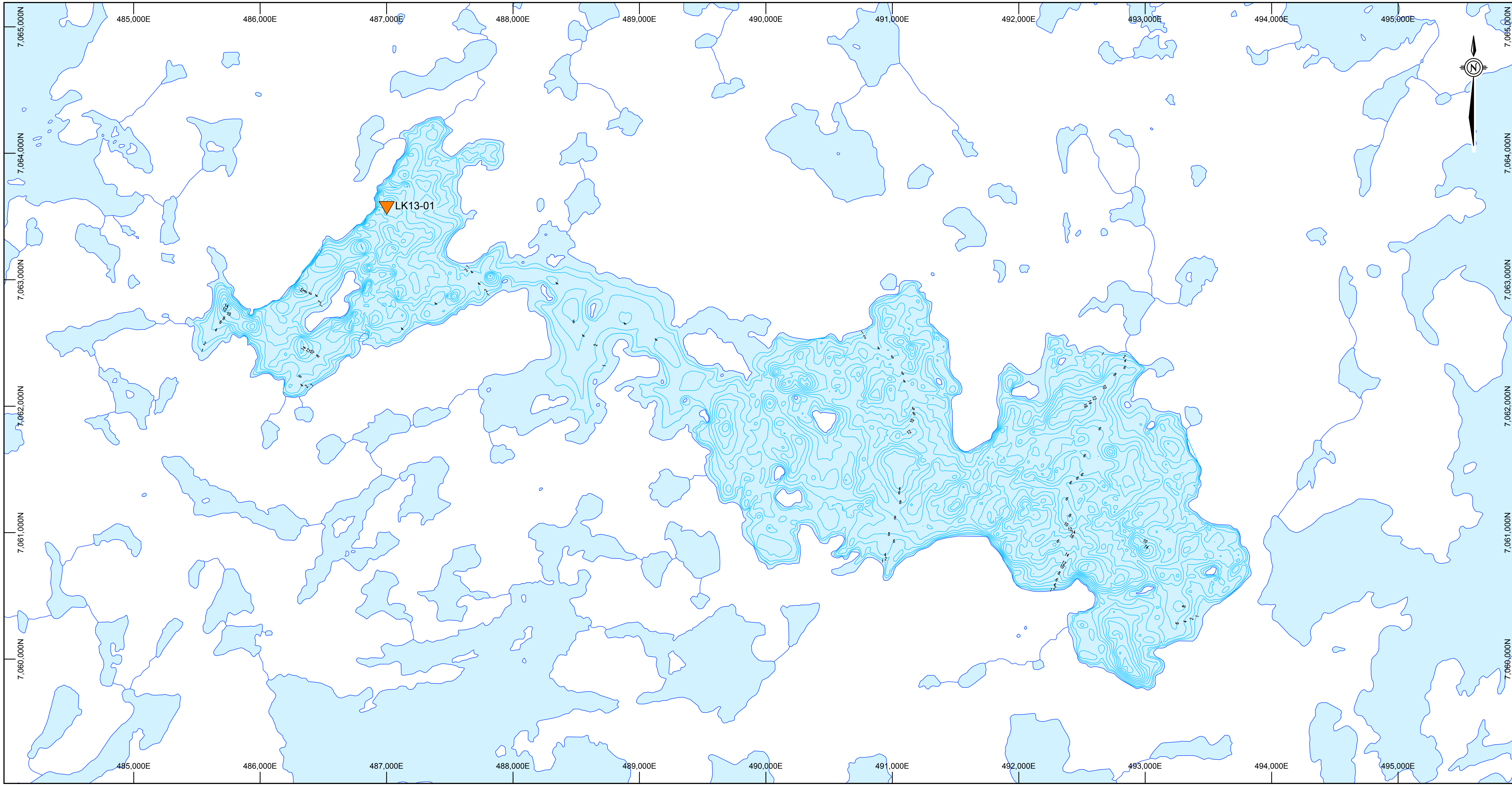


500 0 500

SCALE METRES

L:\2013\1349\13-1349-0001\Phase 4100\ Drawing file: 13134900014100A5-3.dwg Apr 23, 2013 – 12:30pm

LEGEND

PLANKTON MONITORING STATION

WATERBODY

DEPTH CONTOUR (m)

REFERENCE

BASE VECTOR DIGITAL CANVEC 1:50K DATA OBTAINED FROM GEOGRATIS ONLINE.

DATUM: NAD83 PROJECTION: UTM ZONE 12

PROJECT

DE BEERS
GROUP OF COMPANIES

TITLE

PLANKTON MONITORING STATIONS IN PROVISIONAL REFERENCE LAKE 13, 2012

Golder Associates

PROJECT	13-1349-0001.4100	
DESIGN	TD	2013-03-13
CADD	JEF	2013-03-30
CHECK	LMB	2013-04-19
REVIEW	PMC	2013-04-19

FILE No. 13134900014100A5-3
SCALE AS SHOWN REV. 0

FIGURE 5-3

Table 5-1 Summary of Plankton Community Sampling Events in Snap Lake, Northeast Lake, and Lake 13, 2012

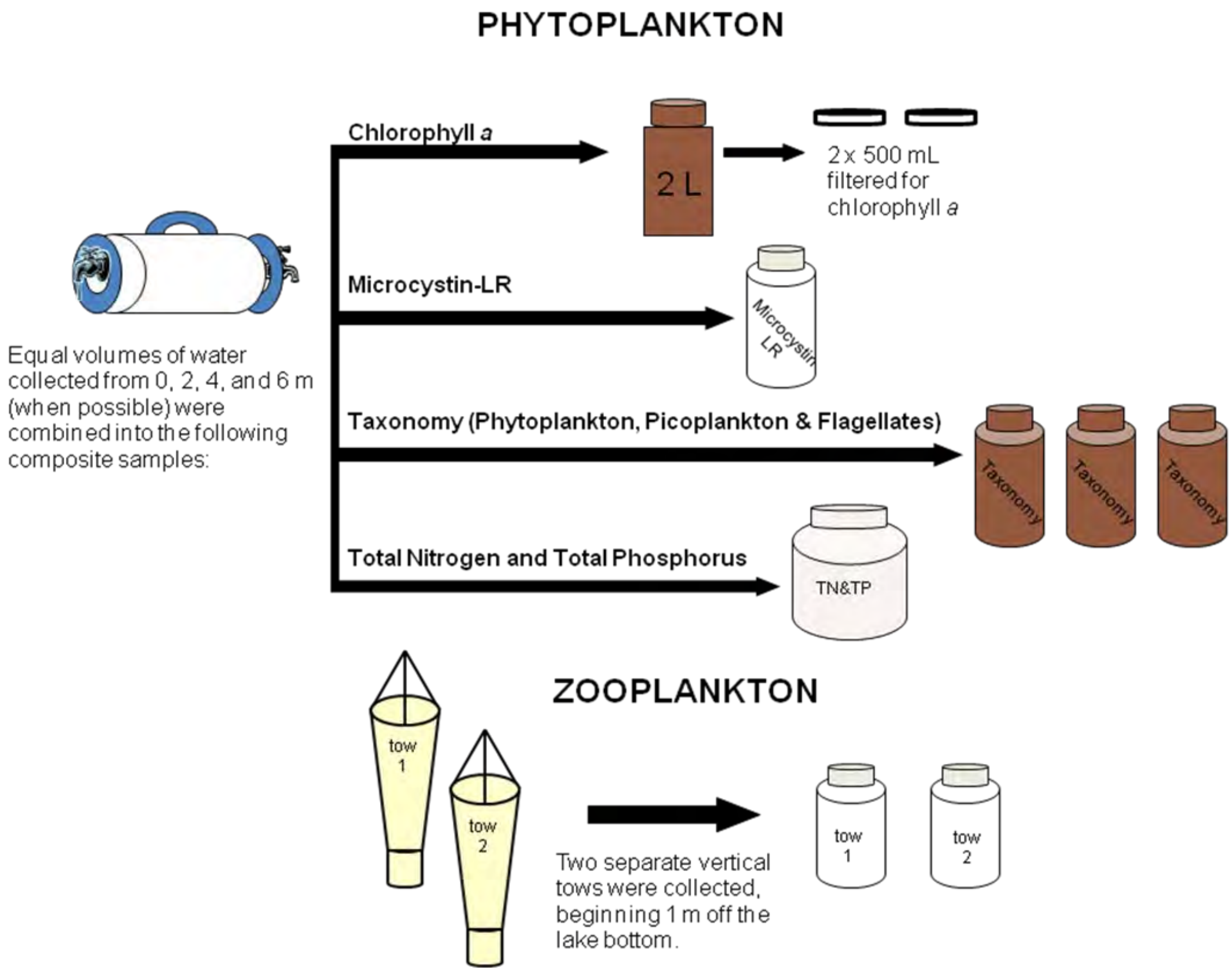
Variable	Trip 1	Trip 2	Trip 3
Snap Lake	**July 7 to 14, 2012 (n)**	**August 10 to 15, 2012 (n)**	**September 7 to 14, 2012 (n)**
Field water quality profile	10	9 [(a)]	10
TN and TP	10	10	10
Phytoplankton	10	10	10
Chlorophyll *a* and *c*[(b)]	18[(c)]	20	20
Microcystin-LR	10	10	10
Zooplankton[(b)]	20	20	20
Picoplankton	10	10	10
Northeast Lake	**July 10, 2012 (n)**	**August 15, 2012 (n)**	**September 8, 2012 (n)**
Field water quality profile	5	5	5
TN and TP	5	5	2 [(d)]
Phytoplankton	5	5	5
Chlorophyll *a* and *c*[(b)]	10	10	10
Microcystin-LR	5	5	5
Zooplankton[(b)]	10	10	10
Picoplankton	5	5	5
Lake 13		**August 8, 2012 (n)**	
Field water quality profile	n/a	0	n/a
TN and TP	n/a	0	n/a
Phytoplankton	n/a	1	n/a
Chlorophyll *a* and *c*[(b)]	n/a	2	n/a
Microcystin-LR	n/a	0	n/a
Zooplankton[(b)]	n/a	2	n/a
Picoplankton	n/a	0	n/a

(a) Field water quality profile data were not collected at SNAP01 in August.
(b) Duplicate samples were collected at each station.
(c) No sample submitted for SNAP08 in July.
(d) No samples submitted for NEL03, NEL04, and NEL05 in September.
n = number of samples; TN = total nitrogen; TP = total phosphorus; - = no sampling event; n/a = not applicable

5.2.1.5 Sampling Methods

A summary of plankton collection methods is presented in Figure 5-4. Detailed collection methods are provided below.

Figure 5-4 Overview of the Plankton Sample Collection Methods



L = litre; mL= millilitre; m = metre; TN & TP = total nitrogen and total phosphorus.

Supporting Environmental Variables

Depth profiles of pH, 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 both Snap Lake, Northeast Lake, and Lake 13. These measurements were collected with a YSI 600-QS multi-meter. Secchi depths were also recorded at each sampling station during each sampling event.

Plankton

In Snap Lake, Northeast Lake, and Lake 13, the upper 6 metres (m) of the water column were sampled at all stations, with the exception of SNAP01, 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 euphotic zone. Water was collected at station SNAP01 from 0 m, 2 m, and 4 m depths. 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*;
- microcystin-LR;
- TN and TP;
- phytoplankton taxonomy; and,
- picoplankton and flagellate taxonomy.

Chlorophyll *a* and *c*: Two composite chlorophyll samples were collected from each station resulting in 20 chlorophyll samples per sampling event in Snap Lake, ten chlorophyll samples per sampling event in Northeast Lake, and two chlorophyll samples in Lake 13 (Table 5-1). Each chlorophyll sample was used to analyze chlorophyll *a* and *c* concentrations. For each chlorophyll sample, 500 or 750 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. This process was repeated, resulting in two samples per station. The filters were frozen and shipped to the University of Alberta Biogeochemical Analytical Laboratory (UofA), in Edmonton, Alberta, where chlorophyll *a* and *c* analyses were completed. One sample from each station was analyzed; the second sample was analyzed only if required, in case of loss of the initial sample or anomalous results that needed to be assessed.

Microcystin-LR: A single composite microcystin-LR sample was collected at each station, resulting in ten microcystin-LR samples per sampling event in Snap Lake and five microcystin-LR samples per sampling event in Northeast Lake (Table 5-1). Microcystin samples were not collected in the initial survey of Lake 13. Samples were kept cool, at 4 degrees Celsius (°C) or frozen, and sent to HydroQual Laboratories in Calgary, Alberta for analyses.

Total Nitrogen and Total Phosphorus: A single composite TN and TP sample was collected at each station, resulting in ten TN and TP samples per sampling event in Snap Lake, and five TN and TP samples per sampling event in Northeast Lake (Table 5-1). Depth-integrated nutrient samples were not collected in the initial survey of Lake 13 because nutrient samples were collected according to methods outlined in the water quality component (Section 3). Samples were kept cool, at 4°C, and shipped to the University of Alberta Biogeochemical Analytical Laboratory for analyses.

Phytoplankton Taxonomy: A single composite phytoplankton sample was collected at each station, resulting in ten samples per sampling event in Snap Lake, five samples per sampling event in Northeast Lake, and one sample 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 taxonomic analyses of species composition, abundance, and biomass.

Zooplankton: Two composite zooplankton samples were collected at each station, resulting in 20 samples per sampling event in Snap Lake, ten samples per sampling event in Northeast Lake, and two samples 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. This process was repeated at each station to collect duplicate samples.

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 taxonomic analyses of species composition, abundance, and biomass.

Picoplankton Special Study - Picoplankton and Flagellate Taxonomy: A single composite picoplankton and a single composite flagellate sample were collected at each station, resulting in ten picoplankton and ten flagellate samples per sampling event in Snap Lake, and five picoplankton taxonomy and five flagellate samples per sampling event in Northeast Lake. Sampling for picoplankton did not occur in Lake 13. Picoplankton are a secondary monitoring component for nutrient enrichment and the initial survey of Lake 13 was focused on comparability of the phytoplankton and zooplankton communities.

Picoplankton and flagellate taxonomy samples were collected in amber Nalgene® bottles to prevent degradation from exposure to light. Picoplankton samples were field-preserved with 2 mL of buffered formalin, while flagellate samples were field-preserved with 2.5 mL of Lugol's solution. Picoplankton taxonomy samples and supporting information (i.e., sample depth and volume) were sent to Advanced Eco-Solutions Inc., Newman Lake, Washington, USA, where heterotrophic bacteria and pico-cyanobacteria were enumerated. Flagellate samples and supporting information (i.e., sample depth and volume) were sent to Eco-Logic Ltd. in Vancouver, British Columbia for analyses, which included identification to the lowest practical taxonomic level (generally species), and calculation of abundance and biovolume for each flagellate taxon.

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 algae. 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 250X 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 50X. An inverted microscope at magnification 200 to 400X 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 using a 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.

Picoplankton: Heterotrophic bacteria and autotrophic pico-cyanobacteria were processed and enumerated using epi-fluorescence microscopy and techniques described by MacIsaac and Stockner (1981, 1993), and Stockner (2005).

Flagellates: Autotrophic, mixotrophic, and heterotrophic nano-flagellates, plus larger dinoflagellates, were enumerated from each sample as part of the Picoplankton Special Study. Prior to quantitative enumeration, the samples were gently shaken for 60 seconds, carefully poured into 25-mL settling chambers, and allowed to settle for a minimum of 6 to 8 hours. Counts were completed by placing the 25-mL settling chambers on a Carl Zeiss Inverted phase-contrast plankton microscope (Utermohl 1958). Between 200 and 250 cells were consistently counted from each sample for statistical accuracy (Lund et al. 1958). The compendia of Prescott (1978), Canter-Lund and Lund (1995), and Wehr and Sheath (2003) were used as taxonomic references.

5.2.2 Data Analyses

This annual plankton AEMP section focused on qualitative assessments of spatial, seasonal, and annual trends in the main basin of Snap Lake, northwest arm of Snap Lake, and Northeast Lake. No quantitative comparisons (i.e., statistical analyses) were completed in 2012; these analyses will be included as part of the Re-evaluation Report in 2015.

Spatial and temporal (i.e., seasonal and year-to-year) trends were evaluated as time-series plots based on station means for the following variables, in addition to the supporting environmental variables:

- nutrients – TN, TP, and the molar ratios of N:P;

- chlorophyll *a* and *c*;
- microcystin-LR;
- phytoplankton abundance, biomass, and taxonomic richness;
- zooplankton abundance, biomass, and taxonomic richness; and,
- picoplankton and flagellate abundances.

Between 2009 and 2012, water samples for nutrient concentrations were collected during the phytoplankton and water quality programs of the AEMP, but the method of collection differed as did the analytical laboratory between the two components. A review of the data indicated differences in the results from the two methods and laboratories. To investigate this difference, and the uncertainty associated with low-level phosphorus measurements, a Nutrient Special Study was completed in 2012 as detailed in Section 12.4 of the present 2012 AEMP Annual Report.

The soluble reactive Si data collected by the water quality component is collected from mid-depth, and plankton samples are depth integrated. All Si data in this section are presented as the annual mean of the mid-column and surface water.

Phytoplankton and zooplankton data were analyzed separately, but the same methods were used to group the data:

- Phytoplankton and zooplankton abundance and biomass data were divided into groups based on taxonomic results.
- Phytoplankton groups were cyanobacteria, chlorophytes, chrysophytes, cryptophytes, dinoflagellates, diatoms, and “others”. The ‘others’ taxonomic groups included euglenophytes and xanthophytes. For the purposes of this study, and to maintain consistency between taxonomy results, haptophytes (when present) were placed with the chrysophytes.
- Zooplankton groups were cladocerans, calanoid copepods, cyclopoid copepods, rotifers, and ciliates.
- 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 variability in community structure between 2004 and 2012.
- Spatial and seasonal comparisons of biomass data were completed for 2004 to 2012 to assess community structure changes. The phytoplankton and zooplankton data sets were separated into three months as outlined in Table 5-2. For 2004 to 2007, these groupings were determined by examining the mean, plus or minus the standard error (±SE), of whole-lake phytoplankton biomass for each sampling event, and grouping together results from similar sampling events. Zooplankton data were grouped in the same way to maintain consistency. For 2008 to 2012, sampling frequency was reduced to once per month during

the open-water season; therefore, each month was considered separately and no pooling of data was required.

Table 5-2 Seasonal Groupings for Evaluating Phytoplankton and Zooplankton Community Composition in Snap Lake, 2004 to 2012

Season	2004	2005	2006	2007	2008	2009	2010	2011	2012
July	mid-July/ early Aug.	mid- to late July	early to mid-July	mid-July	mid-July	mid-July	mid-July	early to mid- July	early to mid- July
Aug.	late Aug./ early Sept.	early to mid- Aug.	late July/ early Aug.	early to mid- Aug.	mid- Aug.	mid- Aug.	mid- Aug.	early to mid- Aug.	early to mid- Aug.
Sept.	late Sept.	late Aug./ early Sept.	mid- Aug./ mid- Sept.	late Aug./ mid- Sept.	early Sept.	mid- Sept.	mid- Sept.	early to mid- Sept.	early to mid- Sept.

Note: Aug.= August; Sept.=September.

Phytoplankton and zooplankton community structures were summarized using the non-parametric ordination method of non-metric multi-dimensional scaling (NMDS; Clarke 1993). Before completing the NMDS, the zooplankton data were log (x+1) transformed. The phytoplankton data were left as untransformed because the transformed data did not improve the separation of the data among stations or years. A Bray-Curtis distance matrix was generated and the NMDS procedure was applied to this matrix. Using rank order information, the relative positions of stations based on community composition was determined. Goodness-of-fit was determined by examining the Shepard diagrams, which are scatter plots of the dissimilarity values calculated from the species data relative to the distances between the sites in the ordination diagram, as well as the stress values, which were calculated from the deviations in the Shepard diagrams (Braak 1995). Lower stress values (i.e., less than 0.2) indicate less deviation and a greater goodness-of-fit (Clarke 1993). Points that fall close together on the NMDS ordination plot represent samples with similar community composition; points that are far apart from each other represent samples with dissimilar community composition. The NMDS was completed using SYSTAT, version 13.0 for Windows (SYSTAT 2009).

5.2.2.1 Approach

The plankton component analyses were designed to answer Key Questions 1 through 5 listed in Section 5.1.2. The Picoplankton Special Study was designed to answer the Key Questions 6 and 7 listed in Section 5.1.2. An overview of the analytical approach associated with each key question is provided in Table 5-3. Specific details relevant to the data analysis methods to address each key question are provided in Table 5-3.

Table 5-3 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 and Northeast Lake?	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 and Northeast Lake, and do these results suggest signs of Mine-related nutrient enrichment or toxicological impairment?	Qualitative comparisons were conducted, comparing the current Snap Lake phytoplankton community to the reference lakes and to baseline (i.e., 2004). NMDS on the phytoplankton biomass data were conducted. This information in combination with the Qualitative Integration Assessment (Section 13) serves to determine whether abundance, biomass, or community composition in Snap Lake show signs of Mine-related nutrient enrichment or toxicological impairment. Quantitative comparisons were completed as part of the 2012 AEMP Re-evaluation (De Beers 2012b). Quantitative comparisons will be completed following three years of data acquisition (i.e., 2015) and will include comparisons to baseline data as well as further temporal and spatial trend analyses.
3. What is the current status, in terms of abundance, biomass and composition, of the zooplankton community in Snap Lake and Northeast Lake, and do these results suggest signs of Mine-related nutrient enrichment or toxicological impairment?	Qualitative comparisons were conducted, comparing the current Snap Lake zooplankton community to the reference lakes and to baseline (i.e., 2004). NMDS on the zooplankton biomass data were conducted. This information in combination with the Qualitative Integration Assessment (Section 13) serves to determine whether abundance, biomass, or community composition in Snap Lake show signs of Mine-related nutrient enrichment or toxicological impairment. Quantitative comparisons were completed as part of the 2012 AEMP Re-evaluation (De Beers 2012b). Quantitative comparisons will be completed following three years of data acquisition and will include comparisons to baseline data as well as further temporal and spatial trend analyses.
4. How do observed changes compare to applicable predictions in the EAR?	A qualitative assessment of annual trends in Snap Lake was conducted and compared to the EAR predictions.
5. How does the plankton community in reference Lake 13 compare to Snap Lake and Northeast Lake? Is reference Lake 13 a suitable reference lake for plankton?	August samples from Snap Lake and Northeast Lake were compared to the August sample from Lake 13 for chlorophyll *a* and *c*, phytoplankton and zooplankton abundance and biomass, and phytoplankton and zooplankton relative biomass. The August NMDS for phytoplankton and zooplankton biomass was also performed using all three lakes to understand how similar Lake 13 is to Snap Lake and Northeast Lake.
6. What is the current status, in terms of abundance, of the picoplankton community in Snap Lake and Northeast Lake, and do these results provide any evidence of Mine-related nutrient enrichment?	A qualitative review of the picoplankton and flagellate data were completed as part of the annual AEMP report. This review evaluated changes in abundance, determined whether there was growth inhibition, and whether this growth inhibition may be related to nutrient enrichment within Snap Lake. Quantitative comparisons were completed as part of the 2012 AEMP Re-evaluation (De Beers 2012b). Quantitative comparisons (i.e., statistical tests) will be completed following three years of data acquisition
7. How do the observed changes in the picoplankton community compare to changes observed in the phytoplankton community?	Qualitative assessment of the spatial and temporal trends observed in the picoplankton, flagellate and phytoplankton communities was conducted.

EAR = Environmental Assessment Report; AEMP = Aquatic Effects Monitoring Program; NMDS: non-metric multidimensional scaling.

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 verify that the data collected were of acceptable quality. In accordance with Golder Associates Ltd. (Golder) QA/QC protocols, all data entered electronically were reviewed for data entry errors and appropriate corrections were made.

Ten percent of both the phytoplankton and zooplankton samples were re-counted by Bio-Limno to verify the taxonomist's counting efficiency. Samples were reanalyzed if 10% or more of these samples were counted incorrectly.

The inherent variability associated with the plankton samples makes the establishment of a QC threshold value difficult. For the purposes of the plankton QC, samples were flagged and assessed further if there was a greater than 50% difference in total abundance between the original and recounted samples.

In addition, the Bray-Curtis index, which is a measure of ecological distance between two communities, was 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. Due to the high variability in species present in the original compared to the recounted samples, the Bray-Curtis comparison tests were performed on species grouped at the major taxa level (i.e., for phytoplankton: cyanobacteria, chrysophytes, chlorophytes, cryptophytes, dinoflagellates, and diatoms; for zooplankton: calanoids, cyclopoids, cladocerans, and rotifers). 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 5-1]

where x_{ik} and x_{jk} are abundance from the original and re-counted samples respectively.

Flagged data were not automatically rejected because of an 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 2011 data for a range comparison. If the data were outside the corresponding 2004 to 2011 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 ten 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 5A tables, but a flag of "X" 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 QA/QC Results

Phytoplankton QC data results indicate that the overall occurrence of dominant species was consistent between the field samples and the QC samples. Five out of the six paired QC samples exceeded a relative percent difference of 50% for total abundance in one or more of the major taxa: SNAP08 (August), SNAP20B (September), NEL01 (July), NEL03 (August), and NEL02 (September; Appendix 5A, Table 5A-6). However, all of the paired QC samples had Bray-Curtis Index values below 0.5 (Appendix 5A, Table 5A-6), indicating a reasonable overall similarity between the paired samples in 2012. Therefore, further follow-up assessments were not performed; it was determined that the 2012 phytoplankton data were of acceptable quality, and no data were invalidated.

The results of the QC check of the zooplankton data indicated that the occurrence of dominant species was consistent between the paired QC samples. All nine paired QC samples exceeded a relative percent difference of 50% for total abundance in one or more of the major taxa: SNAP02-20e (July), SNAP30 (July), SNAP06 (August), SNAP01 (August), SNAP01 (September), SNAP29 (September), NEL01 (July), NEL03 (August), and NEL05 (September). However, all of the paired QC samples had Bray-Curtis Index values below 0.5 (Appendix 5A, Table 5A-6), indicating a reasonable overall similarity between the paired samples in 2012. Therefore, further follow-up assessments were not performed; it was determined that the 2012 zooplankton data were of acceptable quality and no data were invalidated.

As part of the QA/QC process, the chlorophyll data were reviewed for unusually high or low values, and it was determined that the chlorophyll *a* (16.9 µg/L) and chlorophyll *c* concentrations (1.7 µg/L) at SNAP06 were greater than ten times typical lake values. This finding indicated that the results were incorrect, and likely represented a sampling error. Therefore, these values were flagged as outliers and were removed from graphical analyses but were retained in Appendix 5A with an X appended to the data (Table 5A-3).

5.4 RESULTS

Appendix 5A contains detailed results from all sampling events for all components of the plankton program, as follows:

- Appendix 5A, Table 5A-1 – water profile data;
- Appendix 5A, Table 5A-2 – TN and TP concentrations;
- Appendix 5A, Table 5A-3 – chlorophyll *a* and *c* concentrations;
- Appendix 5A, Table 5A-4 – microcystin-LR concentrations;
- Appendix 5A, Tables 5A-5, and 5A-6 – phytoplankton taxonomic data (i.e., abundance, biomass, and QC results);
- Appendix 5A, Tables 5A-7, and 5A-8 – zooplankton taxonomic data (i.e., abundance, biomass, and QC results);
- Appendix 5A, Table 5A-9 – picoplankton (i.e., heterotrophic bacteria and pico-cyanobacteria) enumeration data; and,
- Appendix 5A, Table 5A-10 – flagellate taxonomic data (i.e., abundance and biomass).

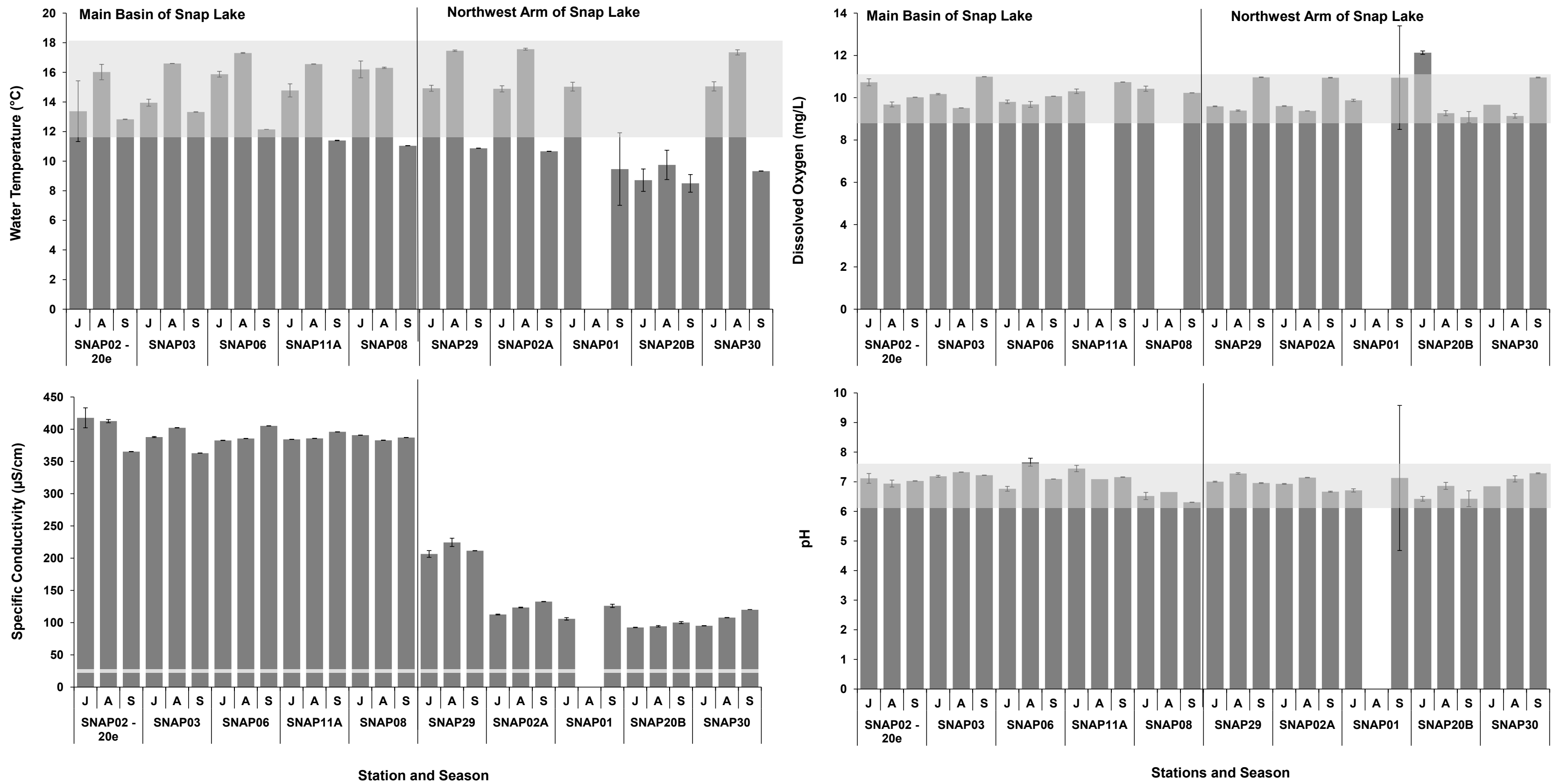
Water profile data, Secchi depths, and nutrient concentrations for Lake 13 are provided and discussed as part of the water quality component (Section 3).

5.4.1 Supporting Environmental Variables

In Snap Lake, the open-water season in 2012 extended from June 10 to October 27, with approximately 140 days of open-water (Section 2). Specific dates of ice-off and lake freeze-up are not available for Northeast Lake; however, the number of open-water days in Northeast Lake is likely comparable to those in Snap Lake, because the lakes are of comparable size and are in close proximity.

Mean water temperatures, based on water-column profile data, were relatively consistent among sampling stations in the main basin of Snap Lake and Northeast Lake in 2012 (Figure 5-5). Mean water temperatures in the northwest arm of Snap Lake were more variable and were often lower than in the main basin of Snap Lake (Figure 5-5).

Figure 5-5 Spatial and Seasonal Trends in Mean Water Temperature, Mean Dissolved Oxygen, Mean Conductivity, and Mean pH Among Plankton Stations in Snap Lake Relative to Northeast Lake, 2012



Note: Stations are arranged from closest (SNAP02-20e) to farthest from the diffuser (SNAP08) in the main basin and northwest arm. Error bars represent standard error of the means. The shaded area represents the data range for each variable in Northeast Lake during the 2012 open-water season. Water temperature, conductivity, pH, and dissolved oxygen were not measured at SNAP01 in August (See Section 5.2.1.4). Dissolved oxygen was also not measured at SNAP11A and SNAP08 in August (Section 5.2.1.4).

J= July; A = August; S = September; mg/L = milligrams per litre; µS/cm = microSiemens per centimeter; and °C = degrees Celsius.

A steady decline in water temperature with increasing depth was observed in July at most stations in the main basin of Snap Lake, and in Northeast Lake (Appendix 5A, Table 5A-1). In August and September, little variation with depth was observed at stations in Snap Lake and Northeast Lake, with the exception of SNAP20B, which was located in the farthest reach of the northwest arm of Snap Lake, and was the deepest station sampled (maximum depth was 37 m but the station was sampled at a depth of 29 m because that was the maximum length of the YSI-cable). Temperatures at SNAP20B decreased by 1.1°C and 0.9°C, with depth, in August and September, respectively, while temperatures at other stations only decreased by an average of 0.2°C during the same months. In September, many stations in Snap Lake had lower mean (±SE) water temperatures (10.9 ± 0.4°C) compared to Northeast Lake (12.9 ± 0.3°C; Figure 5-5). This trend was particularly evident in the northwest arm of Snap Lake, where mean water temperatures at several stations were outside the range observed in Northeast Lake.

The water column remained well oxygenated during all sampling events at most stations in Snap Lake (10.1 ± 0.3 mg/L) and Northeast Lake (10.0 ± 0.1 mg/L; Figure 5-5). Dissolved oxygen concentrations were above the Canadian Water Quality Guideline (CWQG) for non-early life stages of cold water biota (6.5 mg/L; CCME 2007), with the exception of SNAP02A and SNAP30 on August 11, 2012 (6 mg/L at 9 m and 4 mg/L at 10 m), respectively, SNAP03 on July 2, 2012 (5.8 mg/L at 13 m), and SNAP03 on September 7, 2012 (0.9 mg/L at 13 m). The low DO concentrations at all of these stations were measured at the bottom depths, and it is suspected that the YSI probe came into contact with the sediment and resulted in an erroneous measurement, as the overlying water column was well oxygenated (Appendix 5A, Table 5A-1).

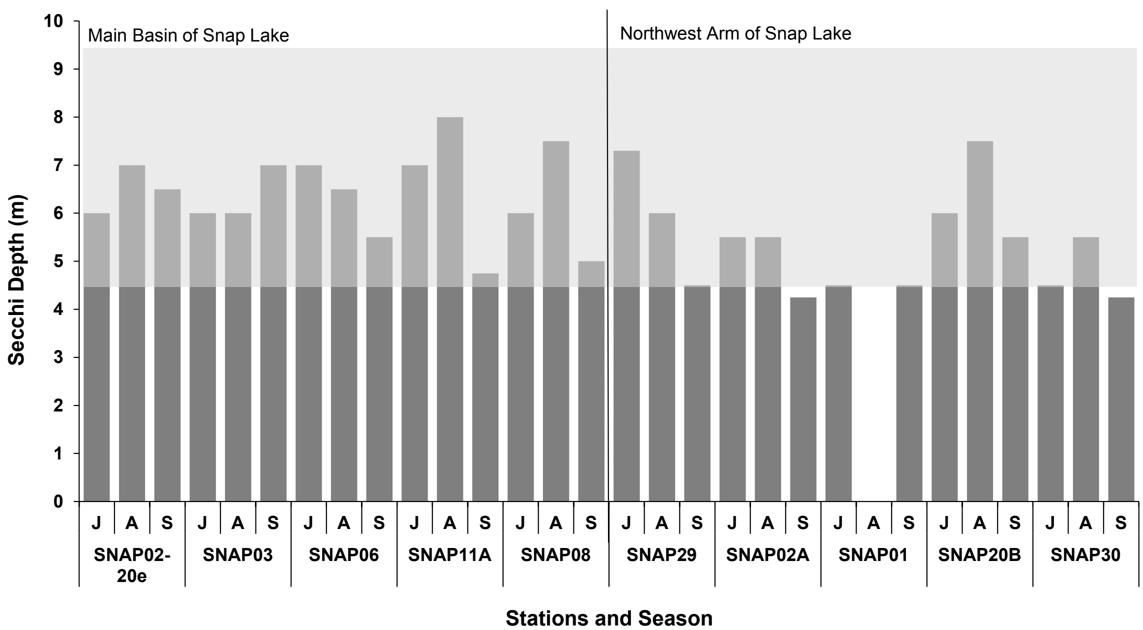
Conductivity in Snap Lake was elevated compared to Northeast Lake. Values were above the baseline level (approximate lake-wide mean of 26 microSiemens per centimetre [µS/cm]; De Beers 2005) in both the main basin and northwest arm (Figure 5-5).

As in previous years, mean (±SE) conductivity was elevated at sampling stations in the main basin of Snap Lake (390 ± 1.4 µS/cm) compared to sampling stations in the northwest arm of Snap Lake (132 ± 1.6 µS/cm), and Northeast Lake (21 ± 0.1 µS/cm; Figure 5-5). As observed from 2007 through 2011 (De Beers 2008, 2009, 2010, 2011, 2012a), conductivity in the northwest arm of Snap Lake was higher in the vicinity of the water intake (SNAP29), than at SNAP01 and SNAP30, which are located farther west in the northwest arm of Snap Lake (Figure 5-5).

The pH values measured in Snap Lake and Northeast Lake were generally neutral (Figure 5.4); however, there were a few instances of mean pH values falling below the CWQG pH range for the protection of aquatic life for cold water biota (pH= 6.5 to 9.0; CCME 2007; Appendix 5A, Table 5A-1). In July, all stations in Northeast Lake had mean (±SE) pH values between 6.2 ± 0.01 and 6.5 ± 0.01. In addition, in the northwest arm of Snap Lake in July, SNAP20B had a mean (±SE) pH of 6.4 ± 0.8. In September, SNAP08 and SNAP20B also fell slightly below water quality guidelines, with mean (±SE) pH values of 6.3 ± 0.05 and 6.4 ± 0.07, respectively. Although other stations (SNAP08, SNAP02-20e, and SNAP20B) had pH values slightly below 6.5 at near bottom depths, the mean pH value for each station remained above 6.5 (Appendix 5A, Table 5A-1).

Secchi depths in the main basin of Snap Lake and Northeast Lake were similar and ranged from 5 to 10 m during the open-water period in 2012 (Figure 5-6). In the northwest arm of Snap Lake, Secchi depths were often lower and ranged from 4 to 8 m. Similar seasonal variability in Secchi depths was observed between both lakes. At most stations in the northwest arm of Snap Lake (SNAP29, SNAP02A, SNAP20B, and SNAP30), in the main basin of Snap Lake (SNAP06, SNAP11A, and SNAP08), and in Northeast Lake, the shallowest Secchi depths were measured in September. The two stations closest to the diffuser in Snap Lake (SNAP02-20e and SNAP03) had the shallowest Secchi depths in July and August. The deepest Secchi depths generally occurred in August in Northeast Lake, with the exception of NEL05.

Figure 5-6 Spatial and Seasonal Trends in Total Secchi Depth Among Plankton Stations in Snap Lake Relative to Northeast Lake, 2012



Note: Stations are arranged from closest (SNAP02-20e) to farthest from the diffuser (SNAP08) in the main basin and northwest arm. Error bars represent standard error of the means. The shaded area represents the data range for each variable in Northeast Lake during 2012. Secchi depths were not measured at SNAP01 in August (See Section 5.2.1.4).

J= July; A = August; S = September; and m = metre.

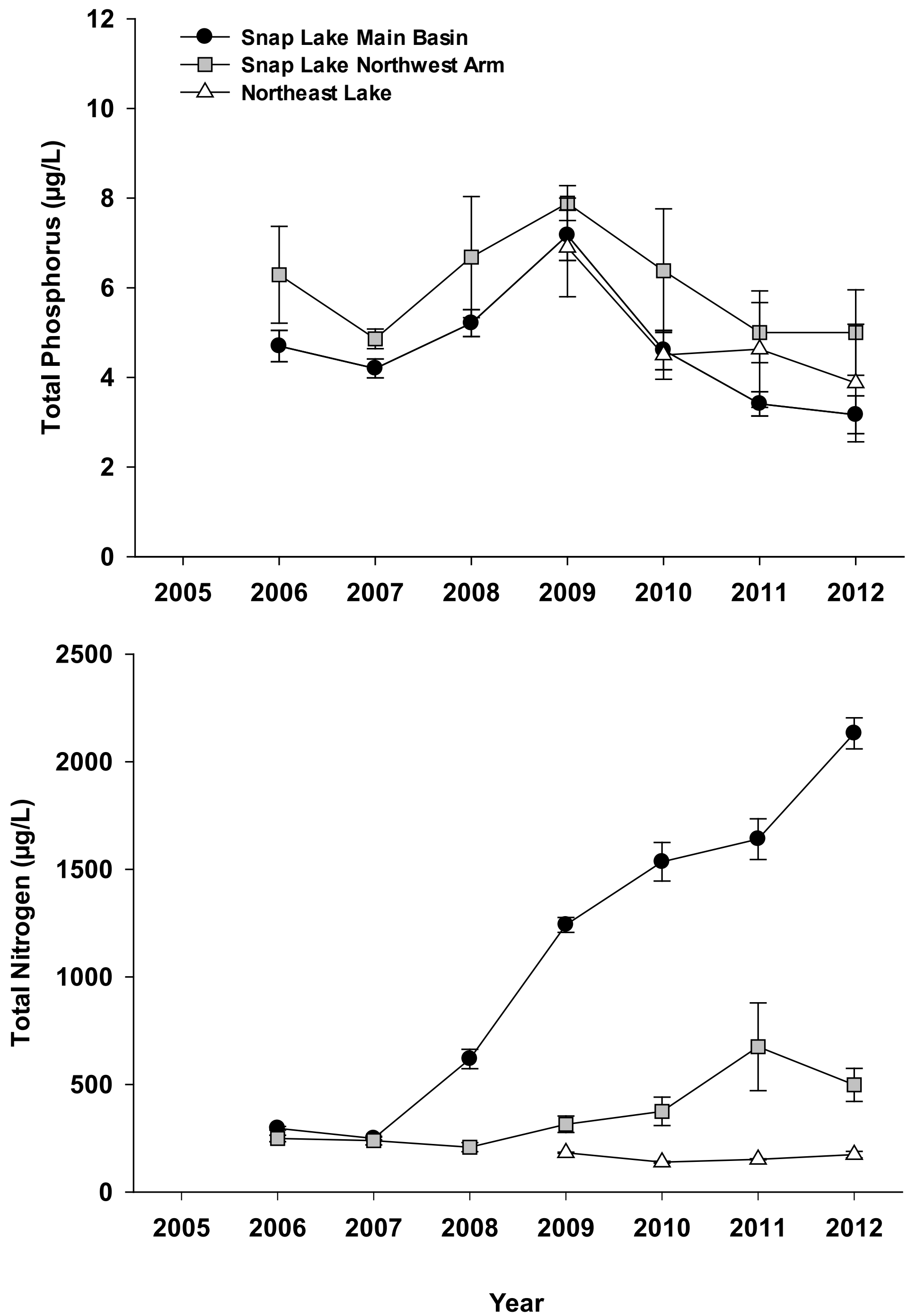
5.4.2 Nutrients

In 2012, the mean (±SE) open-water TP concentration in the main basin of Snap Lake was 3.2 ± 0.4 µg/L (Figure 5-7). The mean (±SE) TP concentration in Northeast Lake (3.9 ± 1.3 µg/L) was comparable to the main basin of Snap Lake, while the mean (±SE) in the northwest arm was slightly higher (5.0 ± 1.0 µg/L). There was one instance of an elevated TP concentration in Northeast Lake in 2012 (NEL04 = 17 µg/L in September), which is uncharacteristically high for this waterbody. Concentrations of TP were elevated in the northwest arm in September, at the two stations located closest to the narrows (12 µg/L at SNAP29 and 15 µg/L at SNAP02A; Figure 5-8; Appendix 5A, Table 5A-2). As in previous years, there was no apparent spatial pattern in TP concentration within Snap Lake relative to proximity to the diffuser (Figure 5-8; Appendix 5A, Table 5A-2).

In 2012, the mean (±SE) TN concentration was lower in the northwest arm of Snap Lake (498 ± 77 µg/L) and Northeast Lake (174 ± 15 µg/L), compared to the main basin of Snap Lake (2,130 ± 72 µg/L; Figure 5-7). In general, stations closer to the diffuser in Snap Lake had higher nitrogen concentrations compared to those further away (Figure 5-9; Appendix 5A, Table 5A-2), with the exception of SNAP03, where the TN concentration was lower (1,360 µg/L) than at SNAP06 (2,110 µg/L) in September. The majority of samples in the northwest arm of Snap Lake had TN concentrations below 1,000 µg/L, except for SNAP29 in July (1,120 µg/L), August (1,060 µg/L), and September (1,010 µg/L; Figure 5-9; Appendix 5A, Table 5A-2). SNAP29 is the station closest to the diffuser in the northwest arm of Snap Lake.

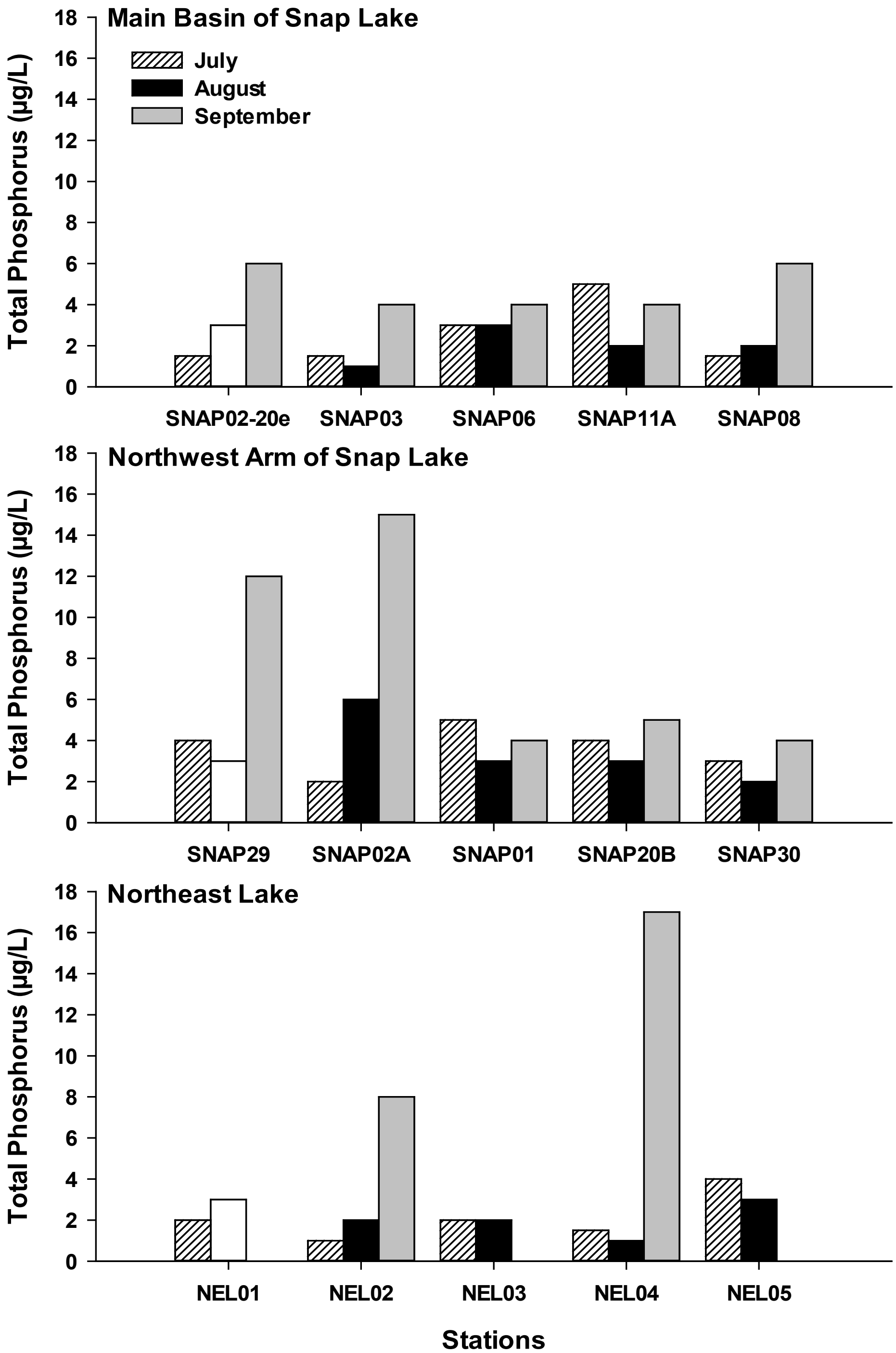
Mean (± SE) Si concentrations were similar between the northwest arm (150 ± 50 µg/L) and the main basin (210 ± 38 µg/L) of Snap Lake in 2005 (Figure 5-10). Since 2005, Si concentrations in the main basin have increased and are now approximately four times higher (Figure 5-10). A similar trend has not been observed in the northwest arm of Snap Lake, where Si concentrations remain close to baseline levels and are comparable to Si concentrations in Northeast Lake. The largest increase in Si concentrations occurred between 2011 and 2012, with mean (± SE) concentrations in the main basin of Snap Lake increasing from 595 ± 17 µg/L to 852 ± 29 µg/L. Although increases in mean Si concentrations were also observed in Northeast Lake and the northwest arm of Snap Lake during the same period, the mean concentrations remained similar to the concentrations measured in 2005.

Figure 5-7 Temporal Trends in Mean Annual Total Phosphorus and Total Nitrogen Concentrations in the Main Basin and Northwest Arm of Snap Lake, 2005 to 2012, and Northeast Lake, 2009 to 2012



Note: Error bars represent standard error of the mean; µg/L = micrograms per litre.

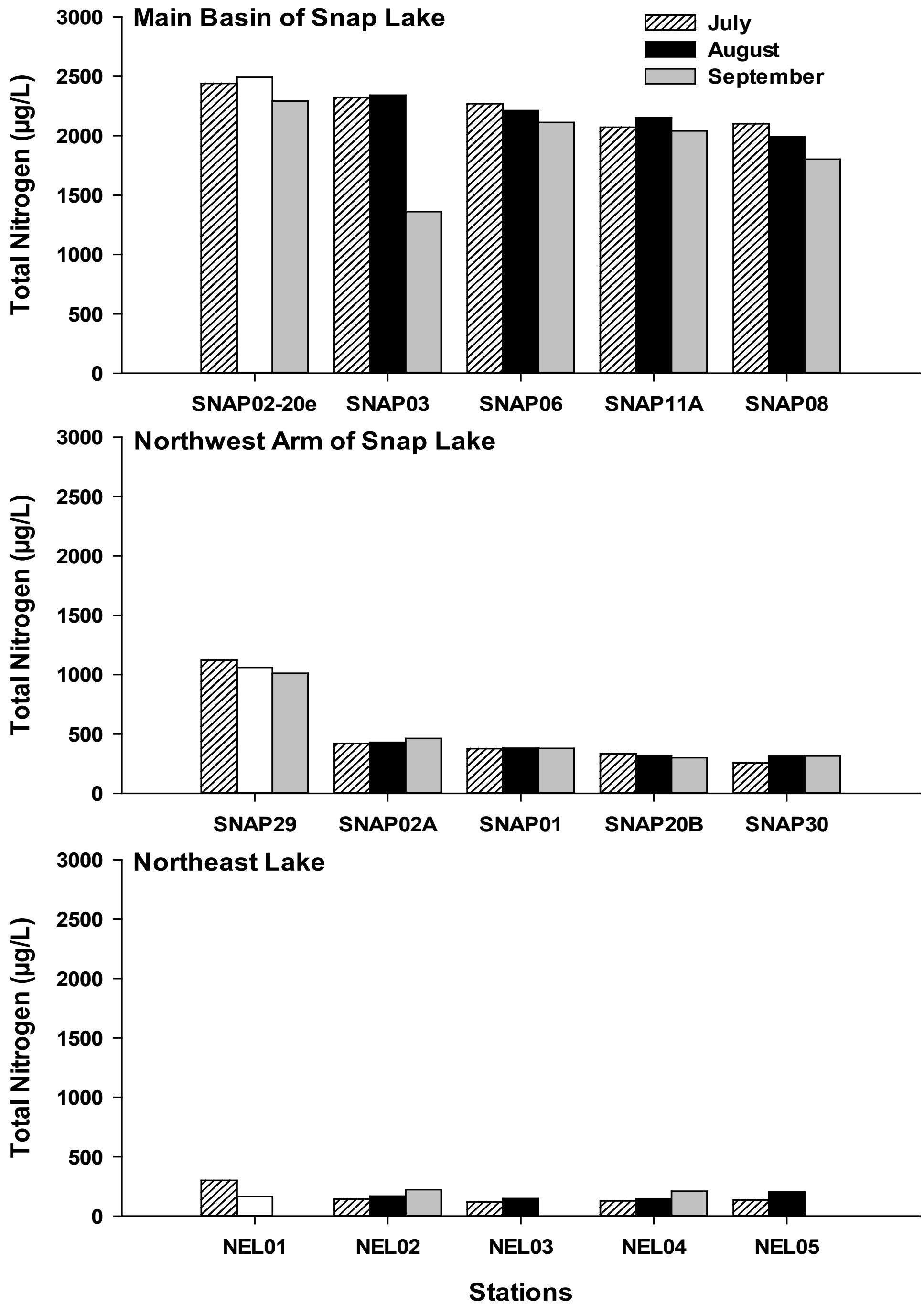
Figure 5-8 Spatial and Seasonal Trends in Total Phosphorus Concentrations in the Main Basin and Northwest Arm of Snap Lake, and Northeast Lake, 2012



Note: Stations are arranged from closest to the diffuser (SNAP02-20e) to farthest from the diffuser (SNAP08) in the main basin and northwest arm of Snap Lake. Total phosphorus samples were not collected at NEL01, NEL03, and NEL05 in September (Section 5.2.1.4). Error bars represent standard error of the mean.

µg/L = micrograms per litre.

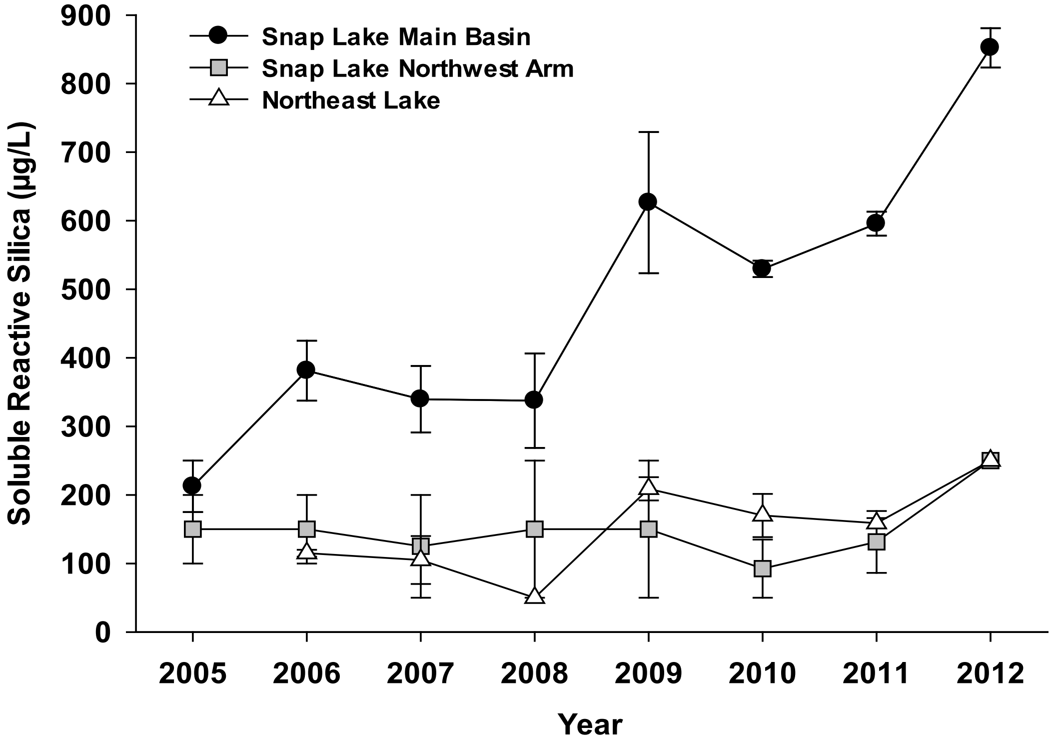
Figure 5-9 Spatial and Seasonal Trends in Total Nitrogen Concentration in the Main Basin and Northwest Arm of Snap Lake, and Northeast Lake, 2012



Note: Stations are arranged from closest to the diffuser (SNAP02-20e) to farthest from the diffuser (SNAP08) in the main basin and northwest arm of Snap Lake. Nitrogen samples were not collected at NEL01, NEL03, and NEL05 in September (See Section 5.2.1.4). Error bars represent standard error of the mean.

µg/L = micrograms per litre.

Figure 5-10 Temporal Trends in Mean Annual Soluble Reactive Silica in the Main Basin and Northwest Arm of Snap Lake and Northeast Lake, 2005 to 2012



Note: Data were collected during the water quality component from mid-depth. Error bars represent standard error of the seasonal mean (n = 2 or 3 depending on the year).

µg/L = micrograms per litre.

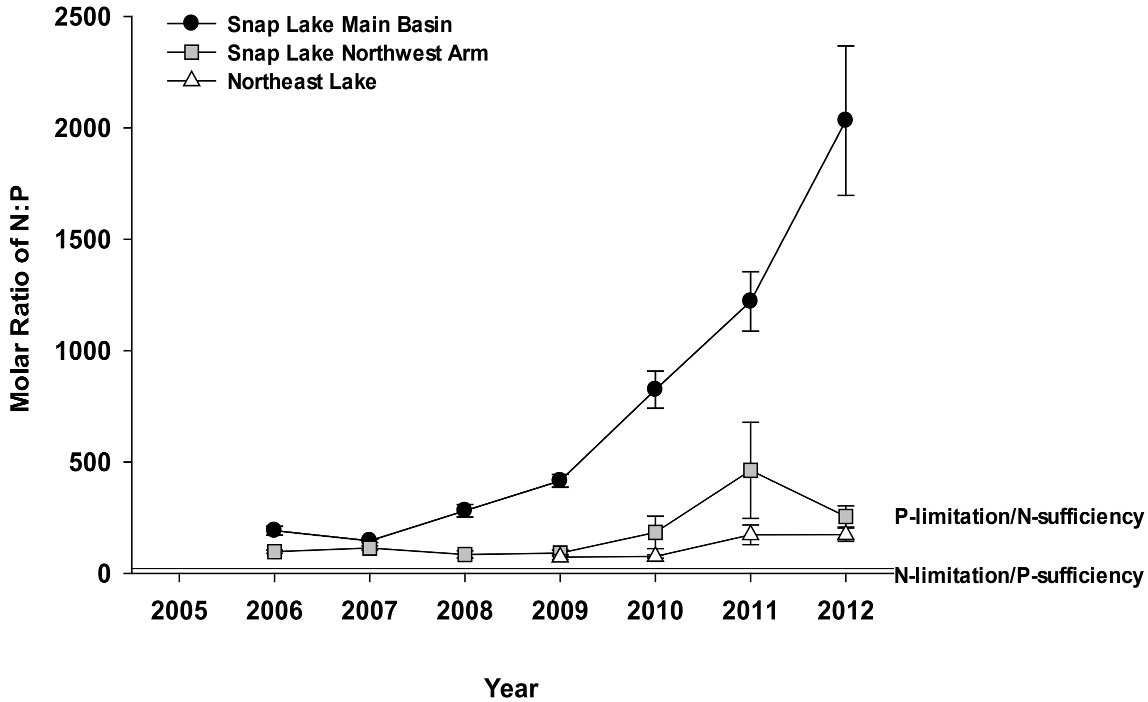
5.4.2.1 Molar Ratio of Nitrogen to Phosphorus

The mean N:P molar ratio has increased steadily in the main basin of Snap Lake since 2007, but has remained relatively unchanged in Northeast Lake and the northwest arm of Snap Lake (Figure 5-11). Variability associated with the mean of the N:P molar ratio has also increased in the main basin of Snap Lake since 2010. This result indicates that P-limitation exists in all of the lakes but is becoming more severe in the main basin of Snap Lake as N concentrations increase.

A seasonal trend in the N:P molar ratio was evident in both Snap Lake and Northeast Lake in 2012 (Figure 5-12). Nearly all stations in both lakes, with the exception of SNAP01 and SNAP11A in Snap Lake, exhibited a decrease in the N:P molar ratio in September. Seasonal peaks in the N:P molar ratio, in both lakes, were observed in either July or August.

In July, stations closer to the diffuser in the main basin had higher N:P molar ratios (3,598 at SNAP02-20e and 3,421 at SNAP03) than all other stations in Snap Lake (Figure 5-12). In addition, SNAP29, the station closest to the diffuser in the northwest arm of Snap Lake, had higher N:P molar ratios in July and August (619 and 781, respectively), compared to the other stations in the northwest arm. This result indicates that P-limitation is more severe in closer proximity to the diffuser as N concentrations increase.

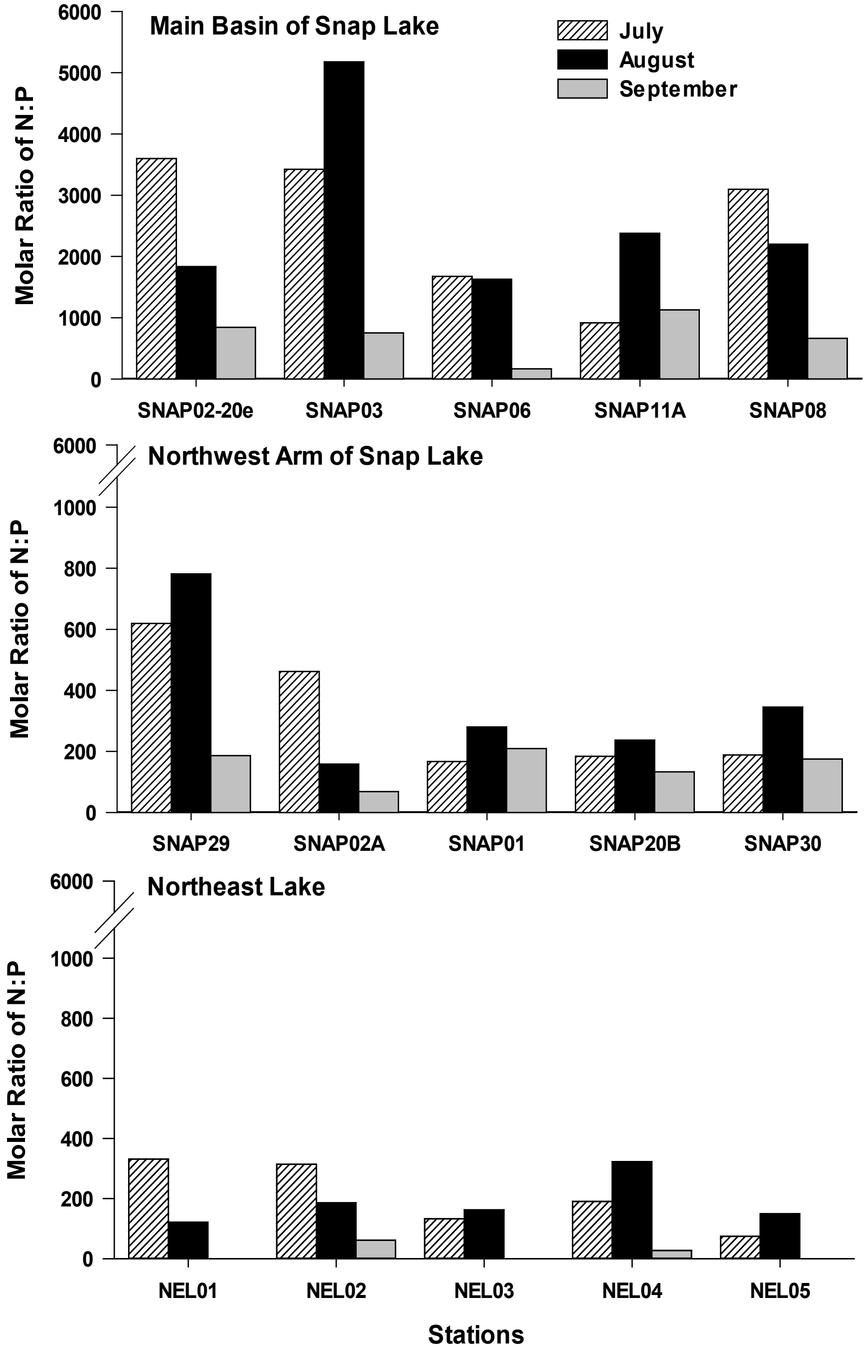
Figure 5-11 Temporal Trends in the Molar Ratio of Nitrogen to Phosphorus in the Main Basin and Northwest Arm of Snap Lake (2005 to 2012), and Northeast Lake (2009 to 2012)



Note: Error bars represent standard error of the mean. The solid horizontal line represents the boundary between nitrogen and phosphorus limitation or sufficiency.

N = nitrogen; P = Phosphorus.

Figure 5-12 Spatial and Seasonal Trends in Mean Open-water Molar Ratios of Nitrogen to Phosphorus in the Main Basin and Northwest Arm of Snap Lake, and Northeast Lake, 2012



Note: Stations are arranged from closest to the diffuser (SNAP02-20e) to farthest from the diffuser (SNAP08) in the main basin and northwest arm. Error bars represent standard error of the mean. Nutrient samples were not collected at NEL01, NEL03, and NEL05 in September (Section 5.2.1.4).

N = nitrogen; P = phosphorus.

5.4.3 Chlorophyll *a* and Chlorophyll *c*

In 2012, chlorophyll *a* concentrations in the main basin of Snap Lake ranged from 0.34 to 16.9 µg/L (Appendix 5A, Table 5A-3). Following removal of the outlier at SNAP06 in August (16.9 µg/L), the mean (±SE) annual chlorophyll *a* concentration in the main basin of Snap Lake (0.9 ± 0.1 µg/L) was consistent with the mean (±SE) concentration observed in Northeast Lake (1.12 ± 0.27 µg/L; Figure 5-13). The mean (±SE) chlorophyll *a* concentration in the northwest arm of Snap Lake (1.65 ± 0.260 µg/L) was higher than those observed in the main basin of Snap Lake and Northeast Lake.

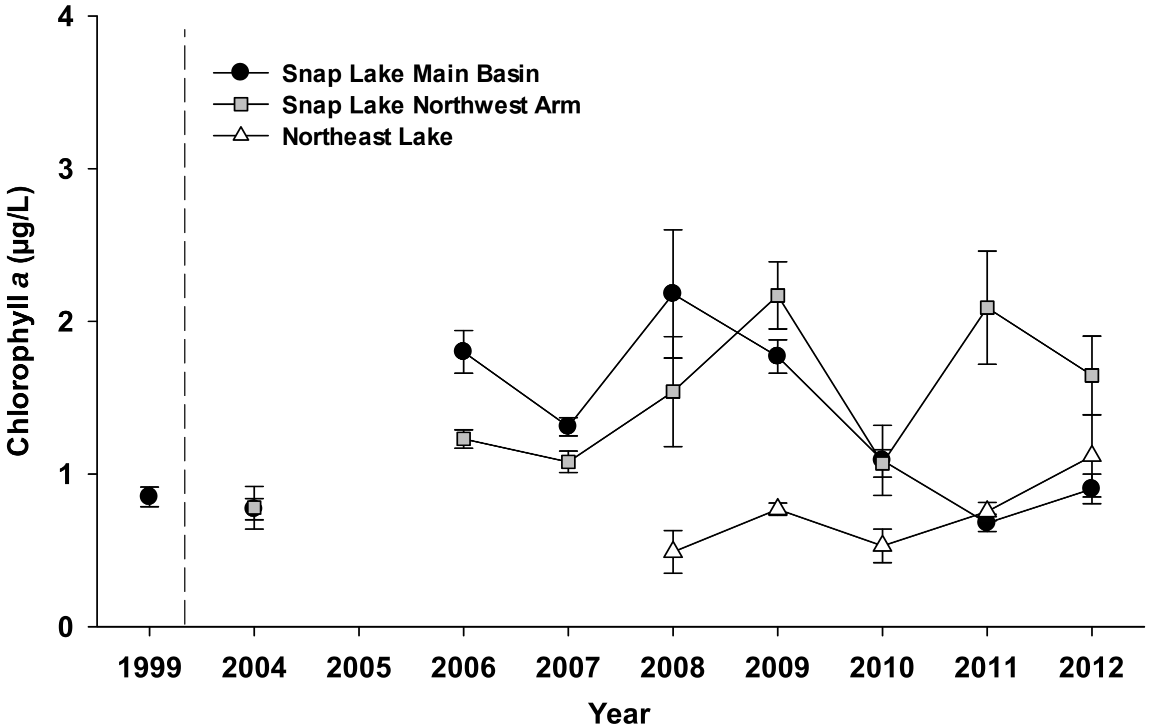
Seasonal trends in chlorophyll *a* concentrations were observed in Northeast Lake and the northwest arm of Snap Lake, with the highest concentrations present in September (Figure 5-14; Appendix 5A, Table 5A-3). Seasonal trends were less evident in the main basin of Snap Lake, where only three of the five stations exhibited peak concentrations in September.

There was no consistent spatial trend in chlorophyll *a* concentrations with proximity to the diffuser in 2012. A weak distance gradient was observed in the main basin of Snap Lake in July, with stations closer to the diffuser (0.64 µg/L at SNAP02-20e) having higher chlorophyll *a* concentrations compared to those farther away (0.13 µg/L at SNAP11A). However, in the northwest arm, stations closer to the diffuser had lower chlorophyll *a* concentrations (0.43 µg/L at SNAP29) compared to those further away (1.5 µg/L at SNAP30).

Chlorophyll *c* concentrations ranged from 0.01 to 2.98 µg/L in Snap Lake, and from 0.03 to 0.54 µg/L in Northeast Lake (Appendix 5A, Table 5A-3). Unusually high chlorophyll *c* values were observed at SNAP30 (2.98 µg/L) in July and SNAP06 (1.7 µg/L) in August. When these outliers were removed, the mean (±SE) annual chlorophyll *c* concentration in the main basin (0.13 ± 0.03 µg/L) was similar to concentrations observed in Northeast Lake (0.15 ± 0.04 µg/L) and the northwest arm of Snap Lake (0.15 ± 0.03 µg/L).

There were no clear seasonal trends in chlorophyll *c* concentrations in Snap Lake. However, in Northeast Lake, chlorophyll *c* concentrations generally peaked in September (Figure 5-15; Appendix 5A, Table 5A-3). Similar to chlorophyll *a*, there was no clear spatial trend in Snap Lake with proximity to the diffuser in 2012 (Figure 5-15; Appendix 5A, Table 5A-3).

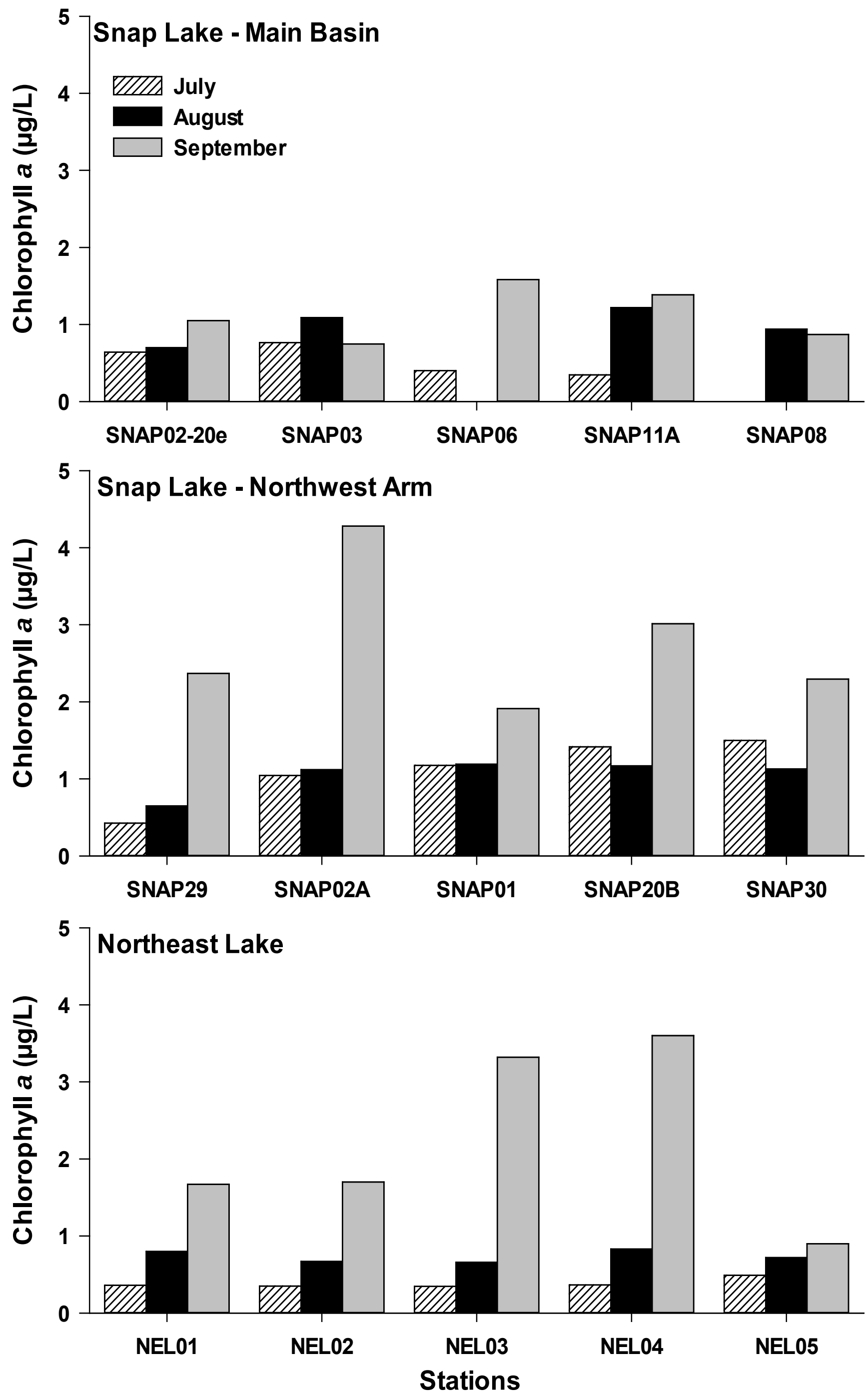
Figure 5-13 Temporal Trends in Mean Annual Chlorophyll *a* in the Main Basin and Northwest Arm of Snap Lake, 1999 and 2004 to 2012, and Northeast Lake, 2008 to 2012



Note: Error bars represent standard error of the means. The 1999 baseline data were not separated into the northwest arm and main basin of Snap Lake (De Beers 2003). Chlorophyll *a* sampling did not occur in Northeast Lake until July 2008 and did not include an August sampling event until August 2011. 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, and data from SNAP06 in August were determined to be outliers and were removed from the plot (Section 5.3.2).

µg/L = micrograms per litre.

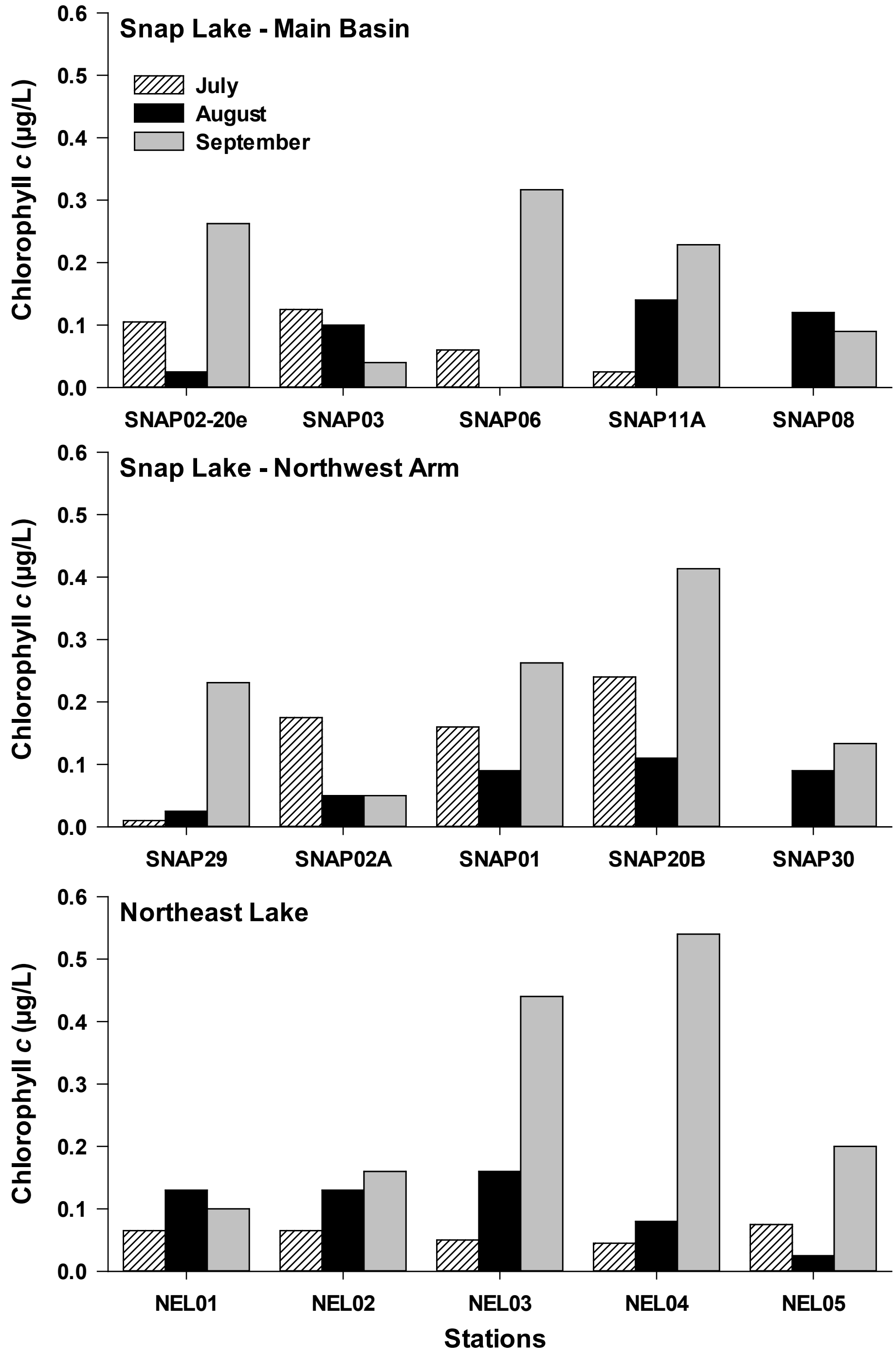
Figure 5-14 Spatial and Seasonal Trends in Chlorophyll *a* Concentrations in the Main Basin and Northwest Arm of Snap Lake, and Northeast Lake, 2012



Note: Stations are arranged from closest to the diffuser (SNAP02-20e) to farthest from the diffuser (SNAP08) in the main basin and northwest arm. Chlorophyll *a* was not collected at SNAP08 in July and data from SNAP06 in August were determined to be outliers and were removed from the plot (Section 5.3.2).

µg/L = micrograms per litre.

Figure 5-15 Spatial and Seasonal Trends in Chlorophyll *c* Concentrations in the Main Basin and Northwest Arm of Snap Lake, and Northeast Lake, 2012



Note: Stations are arranged from closest to the diffuser (SNAP02-20e) to farthest from the diffuser (SNAP08) in the main basin and northwest arm. Chlorophyll *c* was not collected at SNAP08 in July and data from SNAP06 in August were determined to be outliers and were removed from plot (Section 5.3.2).

µg/L = micrograms per litre.

5.4.4 Microcystin-LR

In 2012, most microcystin-LR concentrations in both Snap Lake and Northeast Lake were near or below the detection limit of 0.22 µg/L (Appendix 5A, Table 5A-4). All microcystin-LR concentrations were below the Health Canada (2010) drinking water guideline of 1.5 µg/L.

5.4.5 Phytoplankton

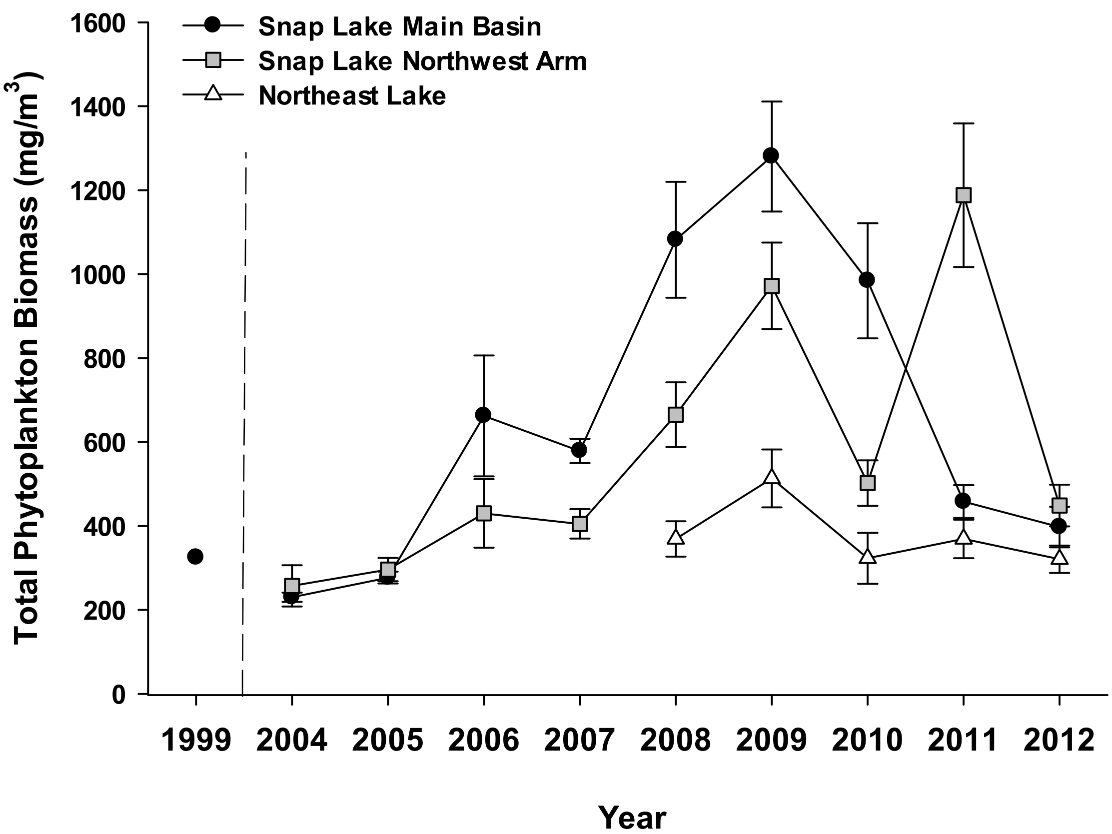
5.4.5.1 Phytoplankton Biomass

Mean annual phytoplankton biomass at stations in the main basin of Snap Lake increased from 2004 to 2009 and then decreased from 2009 to 2012 (Figure 5-16). In 2012, mean (±SE) annual phytoplankton biomass within the main basin was 397 ± 48.0 milligrams per cubic metre (mg/m^3), which was approximately 1.7 times higher than in 2004. Biomass in the northwest arm has generally been lower than in the main basin, but has also shown a steady increase from 2004 to 2011, with the exception of a decrease in 2010 and 2012. Between 2006 and 2010, mean annual phytoplankton biomass in the main basin was higher than in the northwest arm, but became three times lower than in the northwest arm in 2011. By 2012, phytoplankton biomass had become similar between the northwest arm and main basin of Snap Lake, and had returned to near-baseline (2004) levels (230 ±11 mg/m^3). There has been little change in phytoplankton biomass in Northeast Lake since sampling began in 2008 and biomass values have remained close to Snap Lake baseline (1999 and 2004) levels (Figure 5-16).

Seasonal variability in phytoplankton biomass was more pronounced throughout Snap Lake (Figures 5-16 and 5-17) compared to Northeast Lake (Figure 5-19). Peaks in phytoplankton biomass occurred in August in all but one station, SNAP02-20e, in the main basin of Snap Lake. At SNAP02-20e, phytoplankton biomass peaked in July (Figure 5-17). In the northwest arm of Snap Lake, phytoplankton biomass peaked in July at three stations and in August at two stations. In Northeast Lake, phytoplankton biomass peaked in August at all stations except NEL05, which exhibited a peak in September (Figure 5-19).

A spatial trend with proximity to the diffuser was observed in diatom biomass in 2012. With the exception of SNAP02-20e, stations closer to the diffuser had higher diatom biomass compared to stations further away (Figure 5-17). There were no consistent spatial trends observed in any of the major groups in the northwest arm of Snap Lake in 2012 (Figure 5-18).

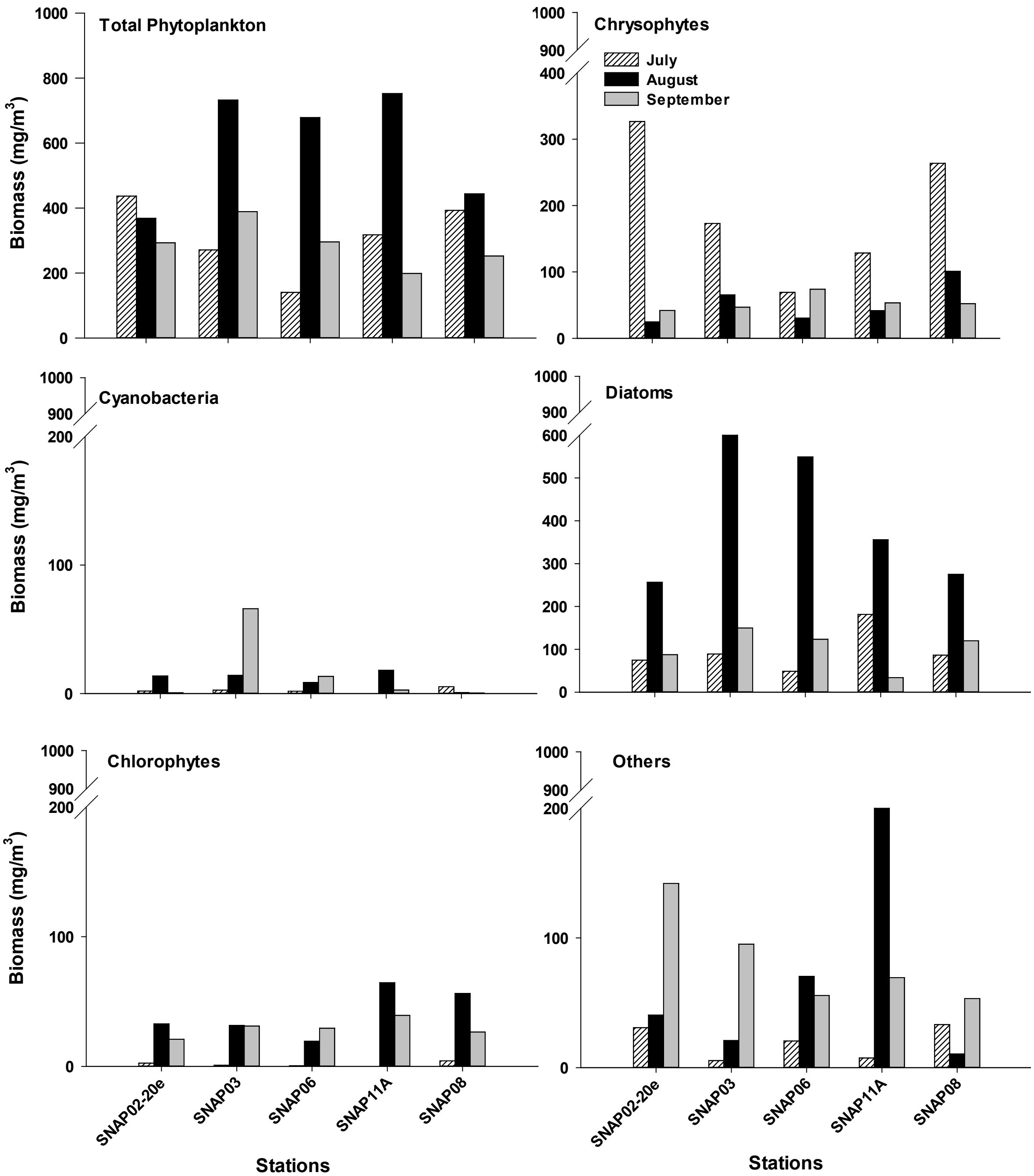
Figure 5-16 Temporal Trends in Mean Annual Phytoplankton Biomass in the Main Basin and Northwest Arm of Snap Lake, 1999 and 2004 to 2012, and Northeast Lake, 2008 to 2012



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 event until 2011. The 1999 baseline data were not separated into the northwest arm and main basin of Snap Lake (De Beers 2003). The vertical dashed bar represents a break in the time series and change in sampling methods.

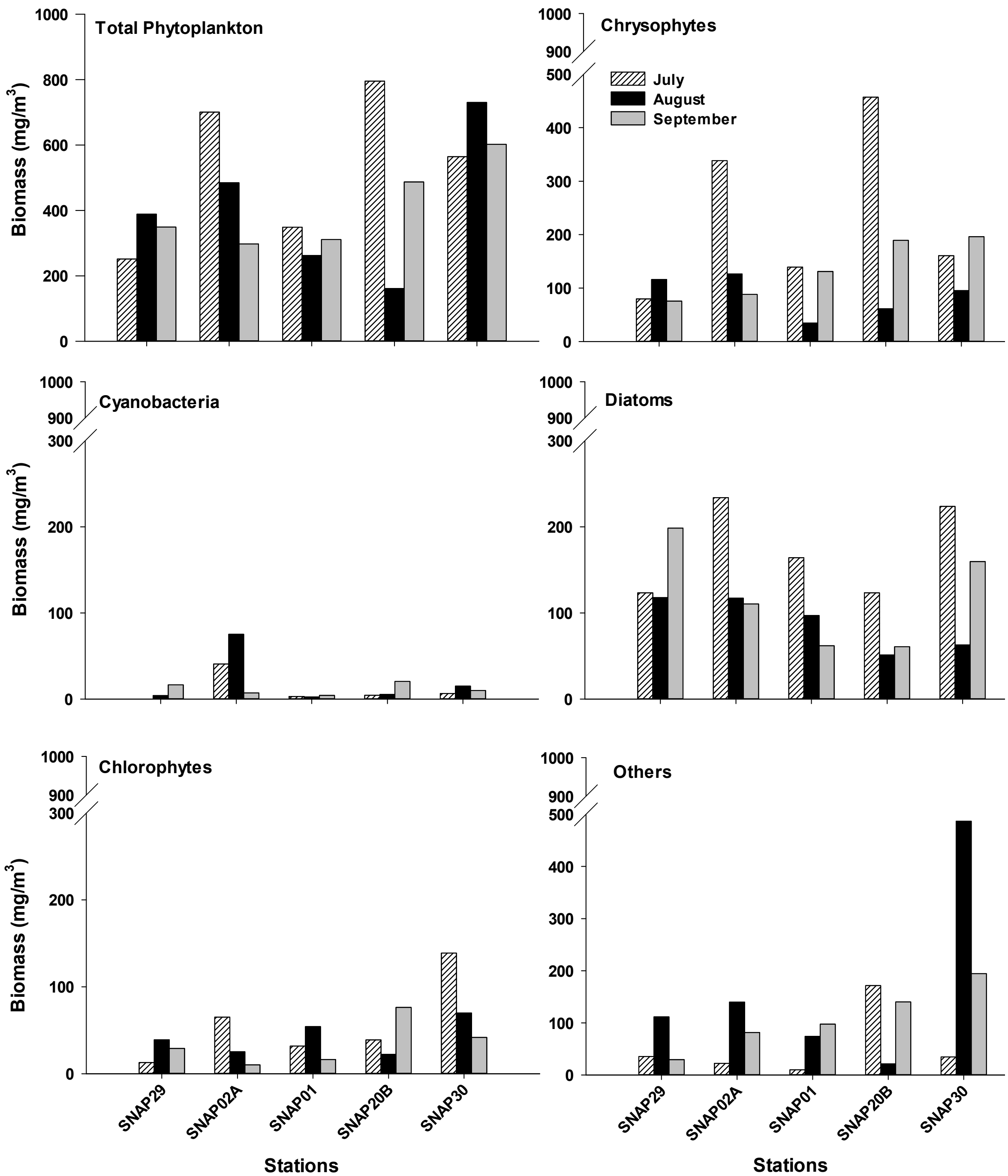
mg/m^3 = milligrams per cubic metre.

Figure 5-17 Spatial and Seasonal Trends in Phytoplankton Biomass in the Main Basin of Snap Lake, 2012



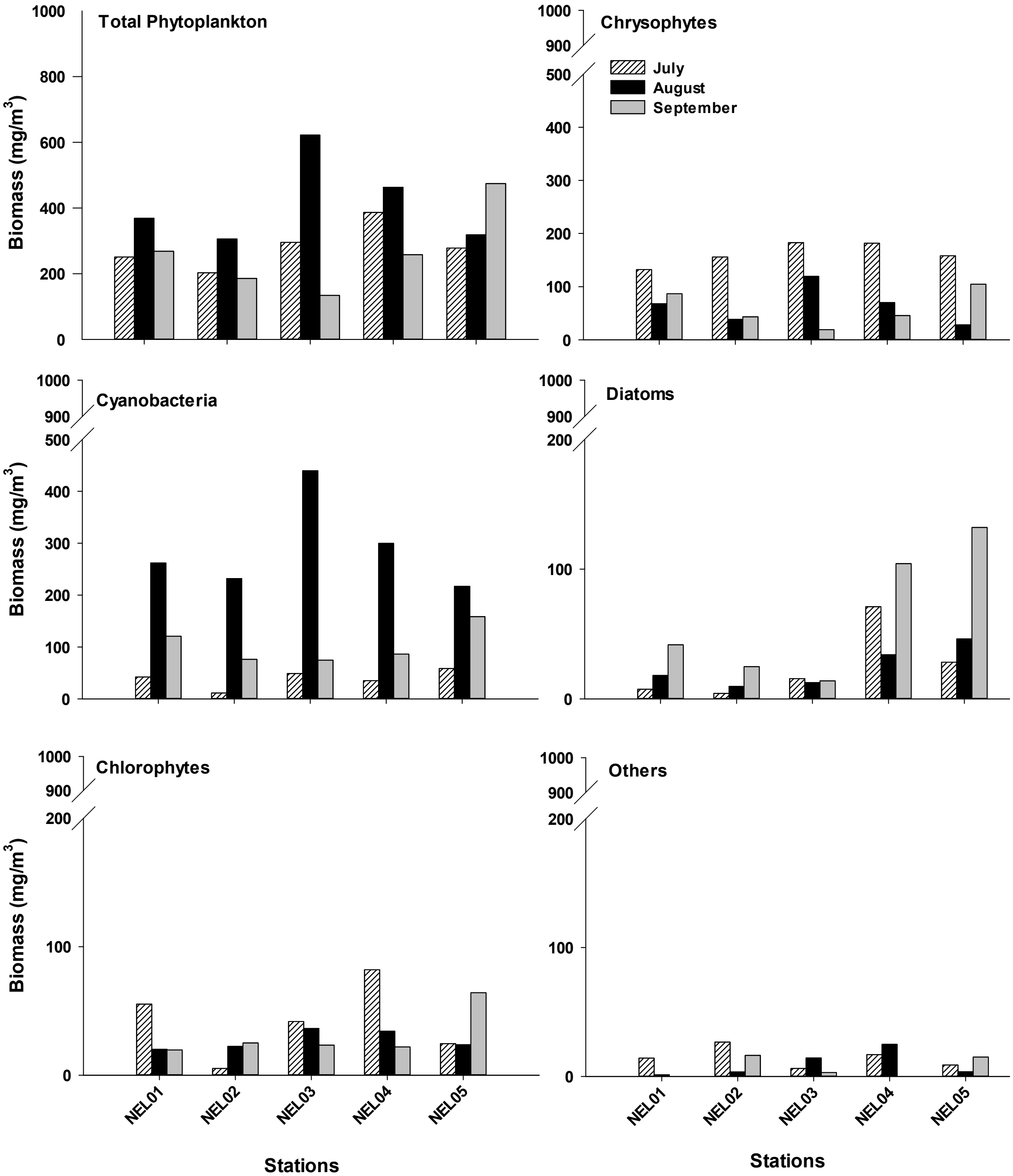
Note: Stations are arranged from closest to the diffuser (SNAP02-20e) to farthest from the diffuser (SNAP08); Others = euglenophytes and xanthophytes; mg/m^3 = milligrams per cubic metre.

Figure 5-18 Spatial and Seasonal Trends in Phytoplankton Biomass in the Northwest Arm of Snap Lake, 2012



Note: Stations are arranged from closest to the diffuser (SNAP29) to farthest from the diffuser (SNAP30)
Others = euglenophytes and xanthophyte;% = percent; mg/m^3 = milligrams per cubic metre.

Figure 5-19 Spatial and Seasonal Trends in Phytoplankton Biomass in Northeast Lake, 2012



Others = euglenophytes and xanthophyte;% = percent; mg/m^3 = milligrams per cubic metre.

5.4.5.2 Phytoplankton Community Composition by Major Taxonomic Group

In the main basin of Snap Lake, chrysophytes and cyanobacteria dominated the plankton assemblage from 2004 to 2006, based on relative biomass (Figure 5-20). In 2007, the relative proportion of cyanobacteria biomass decreased and the community shifted to a diatom-chrysophyte co-dominated community. These two major taxa continued to co-dominate the phytoplankton community biomass in 2012. The relative contribution of chrysophyte biomass to the phytoplankton assemblage in the main basin of Snap Lake varied from 2008 to 2012, while diatom biomass exhibited an increasing trend from 2004 to 2009, and has since comprised between 45% to 60% of the phytoplankton community (Figure 5-20).

In the northwest arm of Snap Lake, phytoplankton community composition, based on relative biomass, has been mainly chrysophyte-dominated, with the relative proportions of cyanobacteria and diatoms varying over time (Figure 5-20). Between 2004 and 2008, the phytoplankton community in the northwest arm of Snap Lake was chrysophyte-cyanobacteria dominated, with the exception of 2006, when there was a relatively high proportion of chlorophytes. Between 2009 and 2012, the phytoplankton community became more evenly distributed among major taxonomic groups, with chrysophytes dominating and sub-dominance shifting among the remaining taxonomic groups. Similar to the main basin of Snap Lake, diatom biomass has increased in the northwest arm of Snap Lake.

Relative abundance of the major taxonomic groups exhibited similar patterns to those observed for relative biomass in Snap Lake (Figures 5-20 and 5-21). The main basin of Snap Lake was chrysophyte dominated from 2004 to 2009, with the exception of 2006, when chlorophytes became the dominant group (Figure 5-21). From 2009 to 2012, diatom abundance increased, and the community shifted to a diatom-chrysophyte co-dominated community.

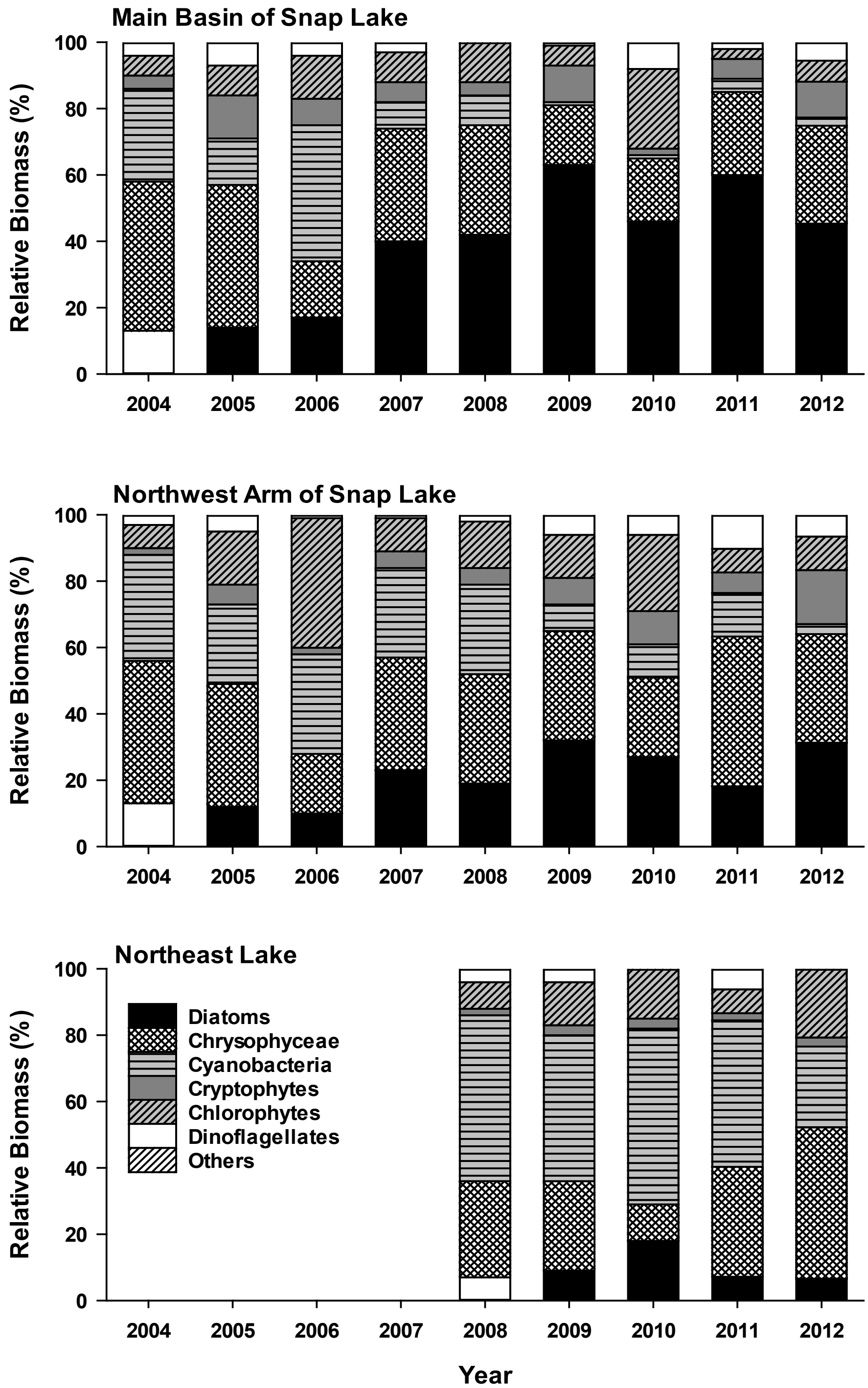
Phytoplankton community composition in Northeast Lake has differed from Snap Lake since sampling began in 2008 (Figures 5-20 and 5-21). Northeast Lake has consistently been a cyanobacteria-chrysophyte dominated lake, by both relative abundance and relative biomass. In 2012, cyanobacterial biomass in Northeast Lake was, on average, approximately ten times higher than in Snap Lake (Figures 5-16, 5-17 and 5-18). Diatom biomass in Northeast Lake has been variable and has not displayed a distinct temporal trend (Figure 5-20).

Chrysophytes have been consistently the dominant phytoplankton group, by relative abundance, in Snap Lake and Northeast Lake since 2004 (Figure 5-21). However, biomass estimates indicate that the relative proportion of chrysophytes has declined, from 51% in 2004 to 40% in 2012, in the main basin of Snap Lake (Figure 5-20). In contrast, diatoms have increased in relative abundance (by 41%) and biomass (by 32%) since 2004 in the main basin (Figures 5-20 and 5-21). In 2012, diatom biomass in Snap Lake was approximately four times greater than diatom biomass in Northeast Lake (Figures 5-17, 5-18, and 5-19).

Dinoflagellates and cryptophytes have consistently accounted for only a small proportion of the Snap Lake phytoplankton community since 2004 (i.e., less than 11% and 17%, respectively; Figures 5-20 and 5-21). Throughout the nine year study period, sporadic increases in the abundance of dinoflagellates occurred at various stations within the northwest arm of Snap Lake. Despite these sporadic increases, there were no discernible spatial or temporal patterns. In Northeast Lake, dinoflagellates and cryptophytes have also accounted for only a small proportion of the phytoplankton community since 2008 (less than 7% and 4%, respectively; Figures 5-20 and 5-21).

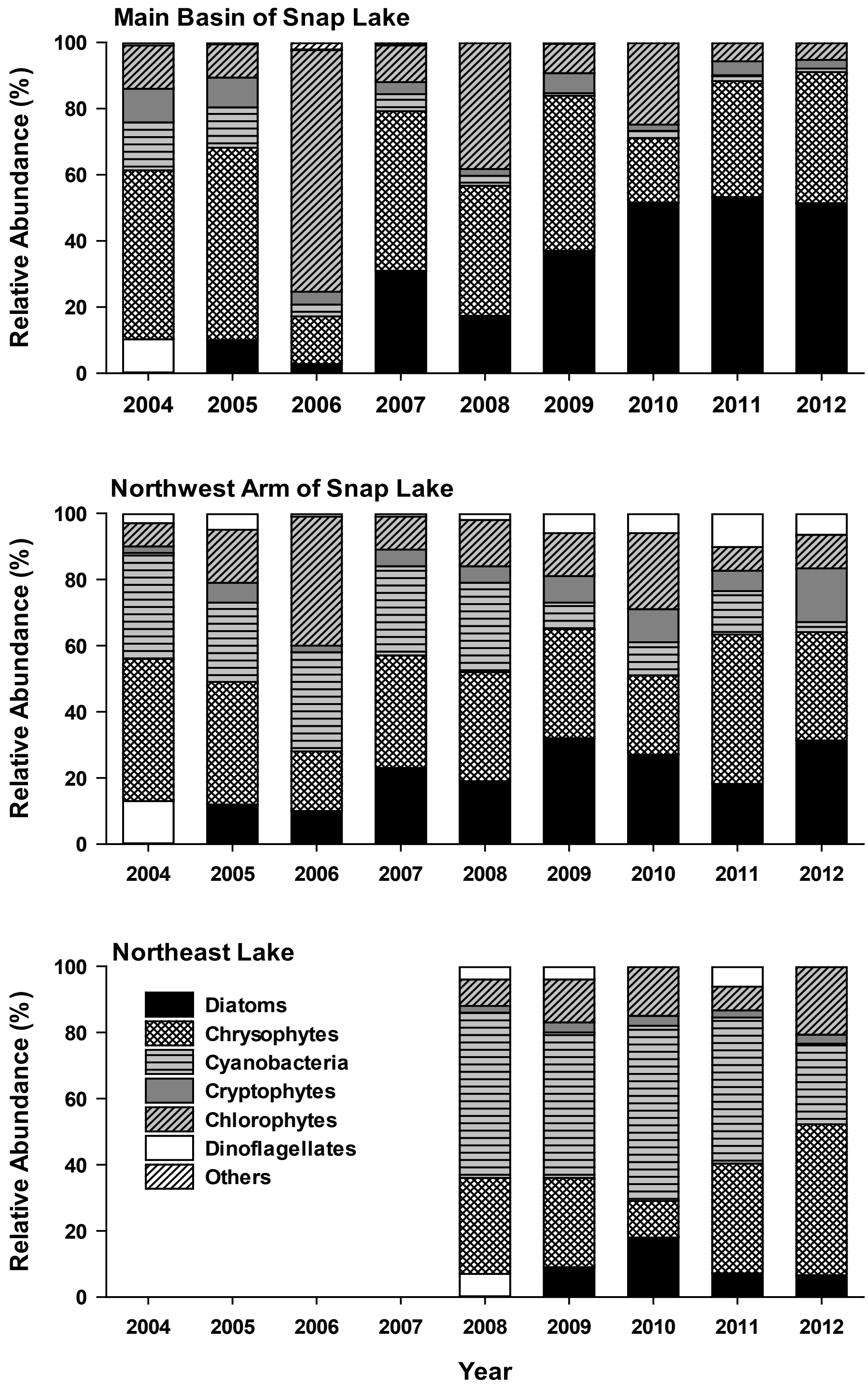
Euglenophytes and xanthophytes have typically been rare groups in Snap Lake. Euglenophytes have been present in Snap Lake since 2005, but have generally accounted for only a small proportion of the phytoplankton community (Figure 5-21). In 2012, euglenophytes were rare in Snap Lake and contributed to less than 1% of the total phytoplankton biomass, and less than 4% of the total phytoplankton abundance (Appendix 5A, Table 5A-5; Figures 5-20 and 5-21). Euglenophytes were not observed in the phytoplankton community in Northeast Lake from 2008 to 2011, and constituted less than 1% of the total phytoplankton biomass and abundance in 2012. Xanthophytes were only present in samples collected from Snap Lake in 2005 (De Beers 2006) and have not been identified in samples from subsequent years.

Figure 5-20 Relative Biomass of Phytoplankton in the Main Basin and Northwest Arm of Snap Lake, 2004 to 2012, and Northeast Lake, 2008 to 2012



Note: Sampling did not occur in Northeast Lake until the July 2008 and did not include an August sampling event until 2011. “Others” were present in small numbers but do not comprise enough biomass to be visible on the plots.
“Others” = euglenophytes and xanthophyte;% = percent.

Figure 5-21 Relative Abundance of Phytoplankton in the Main Basin and Northwest Arm of Snap Lake, 2004 to 2012, and Northeast Lake, 2008 to 2012



Note: Sampling did not occur in Northeast Lake until the July 2008 and did not include an August sampling event until 2011.

“Others” = euglenophytes and xanthophytes;% = percent.

5.4.5.3 Phytoplankton Community Composition by Species

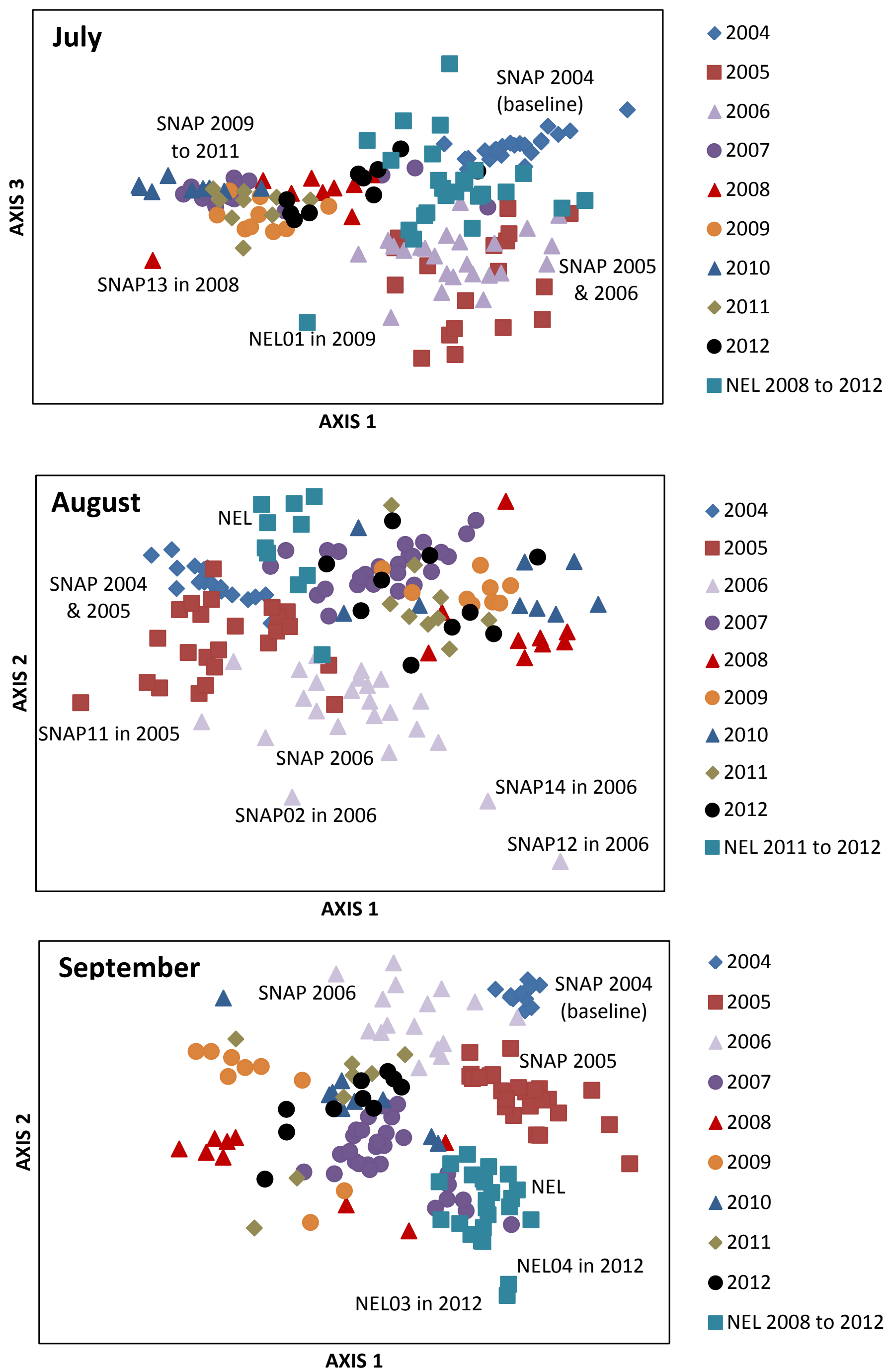
The two-dimensional (August and September) and three-dimensional (July) NMDS configurations showing variation in phytoplankton community composition had stress values of 0.17 (July), 0.22 (August), and 0.20 (September), indicating a reasonable fit to the original data set.

In the July ordination plot, stations in Snap Lake from 2004 were separate from all of the other years (Figure 5-22). Stations in Northeast Lake from 2008 to 2012, grouped closely with the 2004 Snap Lake stations. Snap Lake stations from 2005 and 2006 grouped together but were separate from stations in 2007 and from 2009 to 2011. Stations in Snap Lake from 2008 and 2012 appeared to fall between the 2009 to 2011 cluster and the Northeast Lake and 2004 Snap Lake cluster.

The August ordination plot illustrated that the stations in Snap Lake from 2004 and 2005 grouped closely with stations from Northeast Lake (2011 and 2012). Snap Lake stations from 2007 to 2012 appeared to be randomly distributed throughout the plot. Stations from Snap Lake in 2006 appeared to be grouped separately.

In September, all Northeast Lake stations (2008 to 2012) grouped closely together. Snap Lake data exhibited an orderly progression from 2004 to 2007, but were more randomly scattered from 2008 to 2012.

Figure 5-22 Non-metric Multidimensional Scaling of July, August, and September Phytoplankton Biomass in Snap Lake, 2004 to 2012, and Northeast Lake, 2008 to 2012



5.4.6 Zooplankton

5.4.6.1 Zooplankton Biomass

Mean (±SE) annual zooplankton biomass in the main basin of Snap Lake (70,198 ± 5,549 micrograms per cubic metre [$\mu g/m^3$]) was comparable to Northeast Lake (57,303 ± 4,535 $\mu g/m^3$) in 2012 (Figure 5-23). However, the mean (±SE) biomass in the northwest arm of Snap Lake (126,933 ± 19,254 $\mu g/m^3$) was nearly twice as high as Northeast Lake and the main basin of Snap Lake. Zooplankton biomass was highly variable, showing no clear trend, in the main basin of Snap Lake from 2004 to 2012 (Figure 5-23). Between 2004 and 2007, an increase in mean annual total zooplankton biomass was observed in the main basin of Snap Lake, which was followed by a decrease between 2008 and 2009. Zooplankton biomass increased between 2010 and 2011, but decreased by approximately 50% between 2011 and 2012. Similarly, in the northwest arm of Snap Lake, zooplankton biomass was variable, showing no clear trend. In Northeast Lake, a slight increasing trend was observed between 2008 and 2011, followed by a decrease in 2012.

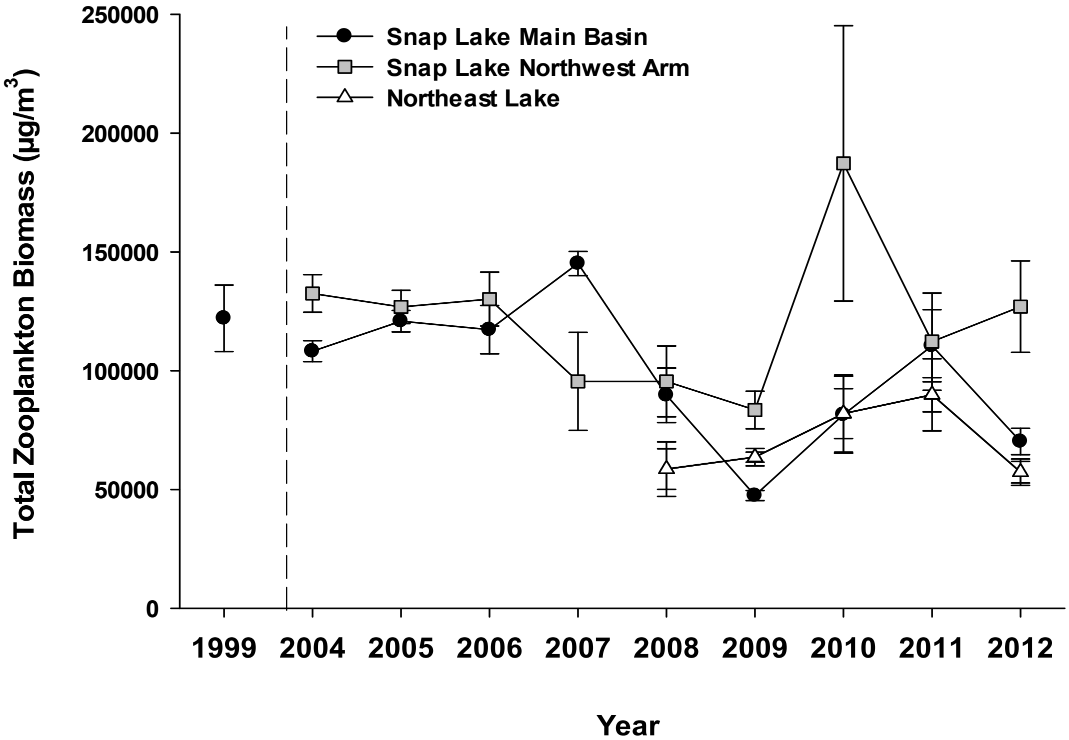
Seasonal peaks in zooplankton biomass differed between the main basin and northwest arm of Snap Lake, as well as between both areas of Snap Lake and Northeast Lake (Figures 5-23, 5-24, and 5-25). In the main basin of Snap Lake, seasonal patterns in zooplankton biomass occurred in August or September at all stations (Figure 5-24). In the northwest arm of Snap Lake, zooplankton biomass peaked in July, except for SNAP20B, where it peaked in August (Figure 5-25). In Northeast Lake, seasonal biomass peaks were observed in August at all stations, except for NEL01, where biomass peaked in July (Figure 5-26).

Northeast Lake exhibited seasonal patterns of greater calanoid copepod biomass in August 2012, compared to July and September 2012 (Figure 5-26). Conversely, Snap Lake exhibited greater calanoid copepod biomass in July (Figure 5-24 and 5-24). Calanoid copepod biomass was similar at most stations in Snap Lake during the July sampling period (37,315 to 64,171 $\mu g/m^3$), with the exception of SNAP02-20e (10,043 $\mu g/m^3$) and SNAP20B (15,936 $\mu g/m^3$), where calanoid copepod biomass was lower (Figure 5-24). In August and September, calanoid copepod biomass was lower in the northwest arm of Snap Lake compared to most stations in the main basin (Figures 5-24 and 5-25).

Cladoceran biomass generally peaked in August and rotifer biomass peaked in July in the northwest arm of Snap Lake (Figure 5-25). No seasonal patterns were observed in cladoceran biomass in the main basin of Snap Lake or Northeast Lake (Figures 5-24 to 5-26). Rotifer biomass peaked in September in the main basin of Snap Lake (Figure 5-25). Cyclopoid copepod biomass did not display clear seasonal patterns in either the main basin or northwest arm of Snap Lake or Northeast Lake (Figures 5-24 to 5-26).

There was a slight increasing trend in calanoid copepod biomass with increasing distance from the diffuser in the main basin in 2012 (Figure 5-24). Otherwise, there were no clear spatial patterns evident in the main basin or northwest arm of Snap Lake in 2012 (Figures 5-24 and 5-25).

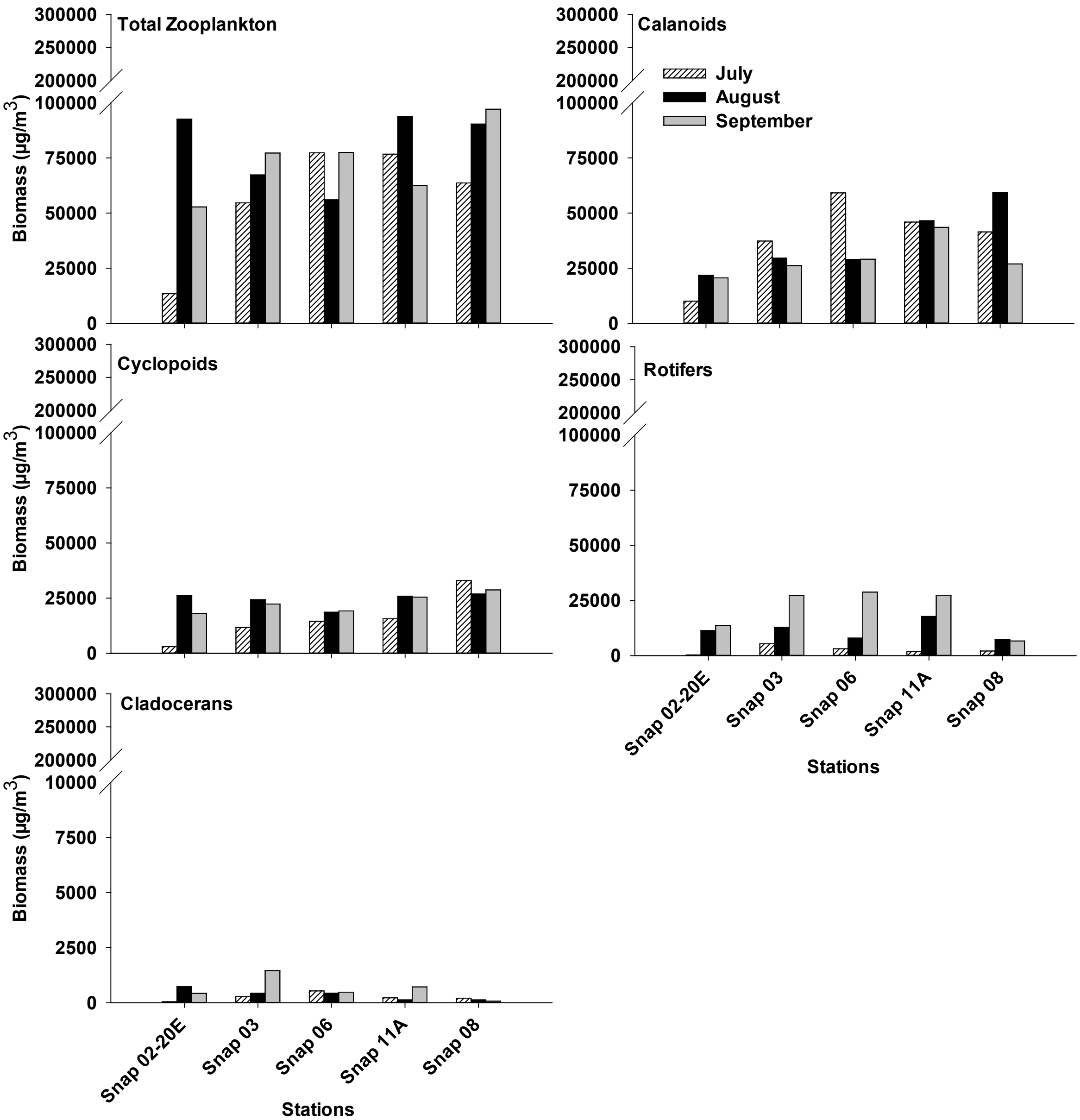
Figure 5-23 Temporal Trends in Mean Annual Zooplankton Biomass in the Main Basin and Northwest Arm of Snap Lake, 1999 and 2004 to 2012, and Northeast Lake, 2008 to 2012



Note: Error bars represent standard error of the mean. . The 1999 baseline data were not separated into the northwest arm and main basin of Snap Lake (De Beers 2003). Sampling did not occur in Northeast Lake until July 2008 and did not include an August sampling until 2011. The vertical dashed bar represents a break in the time series and change in sampling methods.

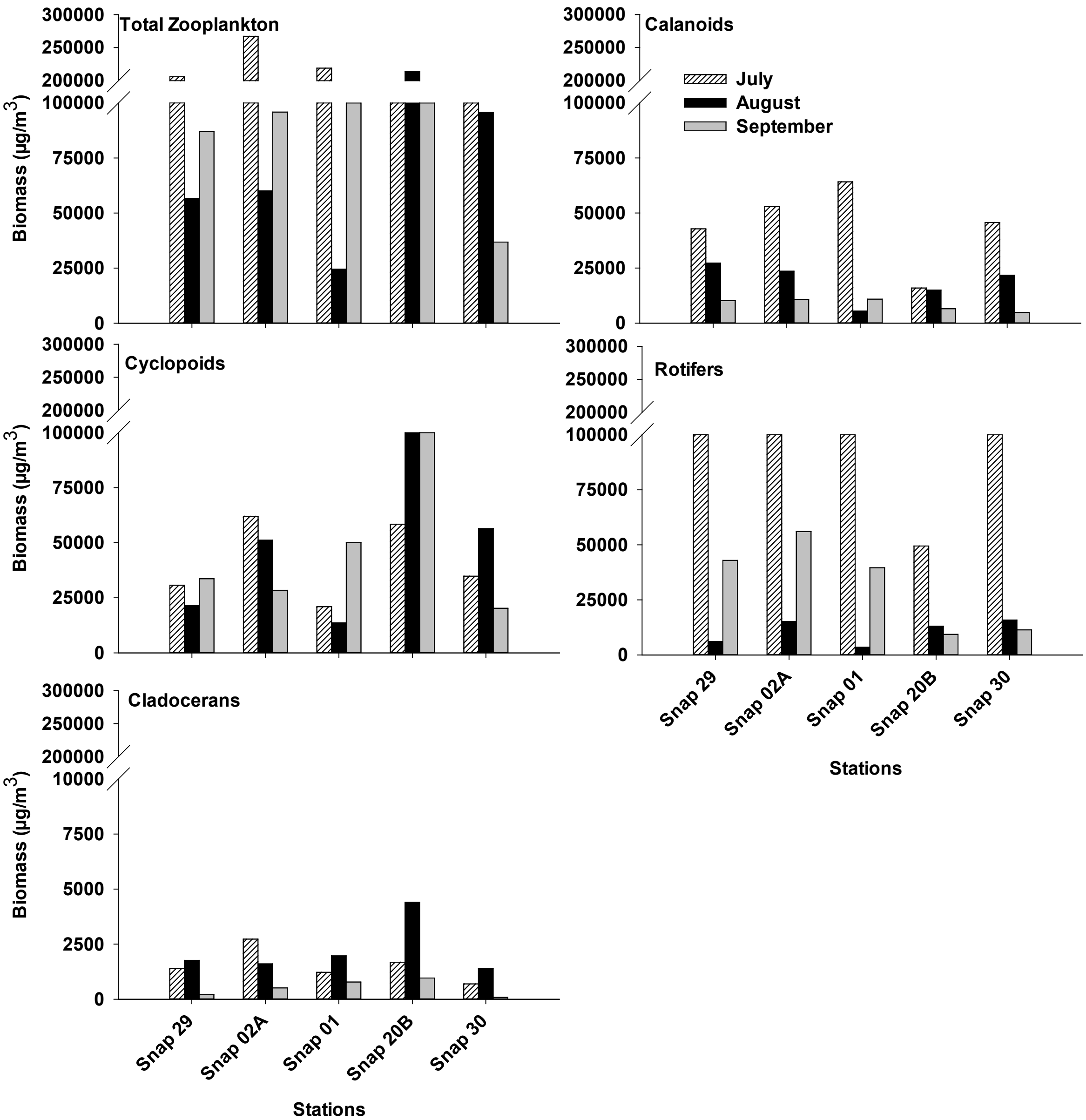
µg/m^3 = micrograms per cubic metre.

Figure 5-24 Spatial and Seasonal Trends in Zooplankton Biomass in the Main Basin of Snap Lake, 2012



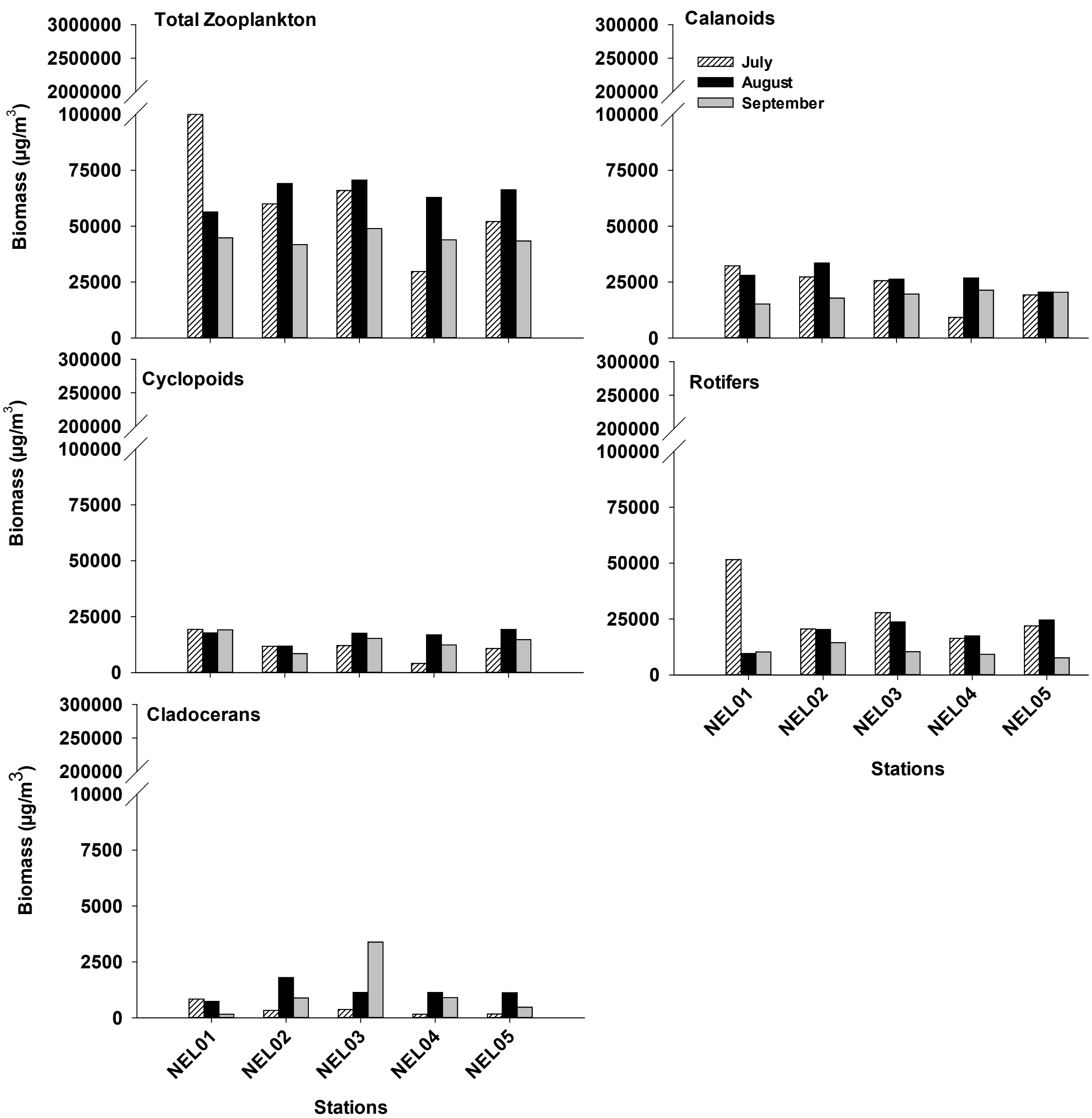
Note: Stations are arranged from closest to the diffuser (SNAP02-20e) to farthest from the diffuser (SNAP08).
µg/m^3 = micrograms per cubic metre.

Figure 5-25 Spatial and Seasonal Trends in Zooplankton Biomass in the Northwest Arm of Snap Lake, 2012



Note: Stations are arranged from closest to the diffuser (SNAP29) to farthest from the diffuser (SNAP30).
μg/m^3 = micrograms per cubic metre.

Figure 5-26 Spatial and Seasonal Trends in Zooplankton Biomass in Northeast Lake, 2012



µg/m³ = micrograms per cubic metre.

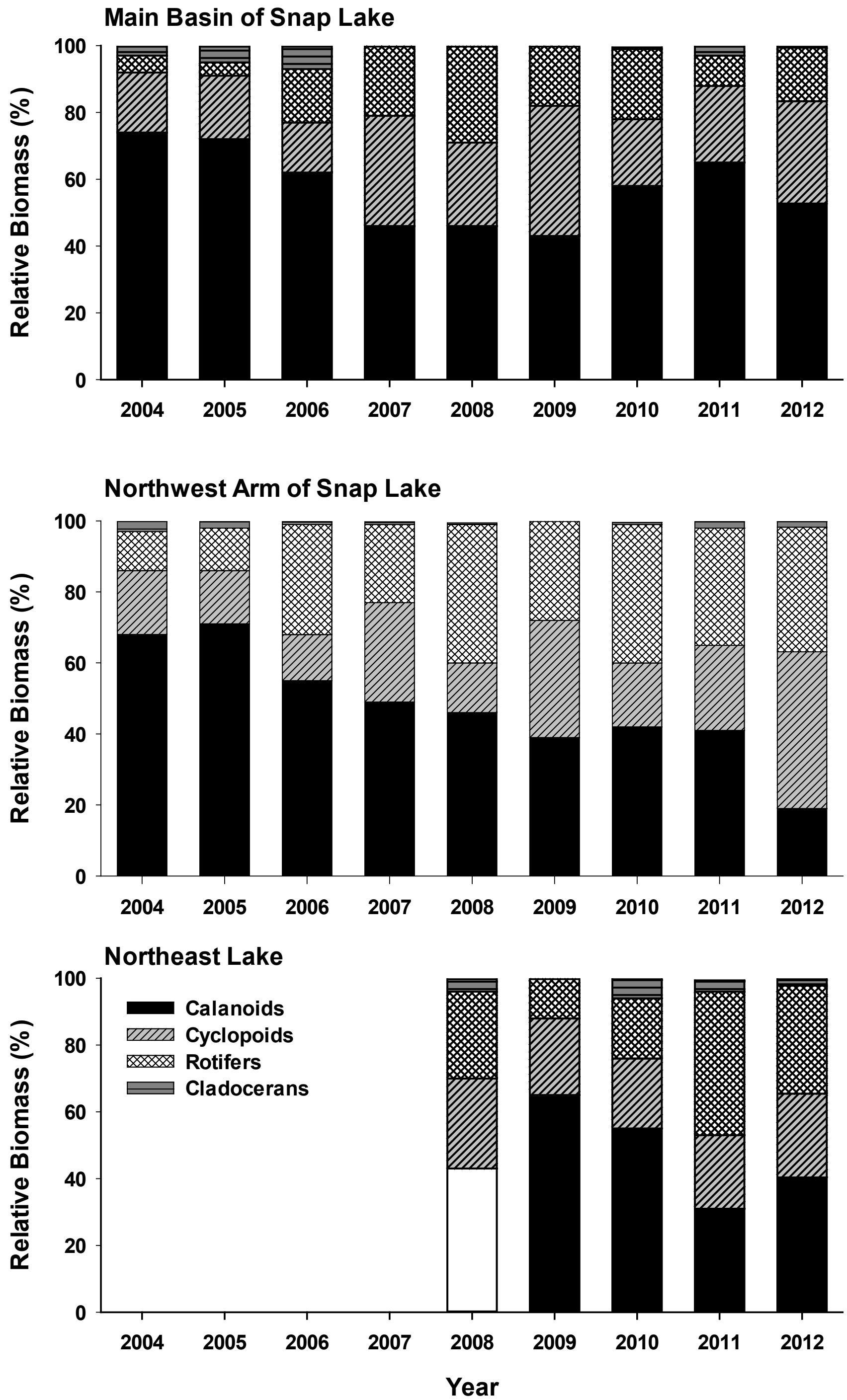
5.4.6.2 Zooplankton Community Composition by Major Taxonomic Group

From 2004 to 2012, relative zooplankton biomass in the main basin of Snap Lake has been consistently dominated by calanoid copepods (Figure 5-27), with cyclopoid copepods or rotifers as the sub-dominant group. Conversely, zooplankton abundance in the main basin has shifted from calanoid copepod dominance in 2004 and 2005, to rotifer dominance from 2006 to 2010, to a calanoid-cyclopoid-rotifer co-dominated community in 2011 and 2012 (Figure 5-28).

In the northwest arm of Snap Lake calanoid copepods have shown a decrease in dominance since 2004, in both abundance and biomass (Figures 5-27 and 5-28). In contrast, the relative proportion of rotifer and cyclopoid copepod biomass has increased in the northwest arm of Snap Lake; only the relative proportion of rotifers increased in abundance.

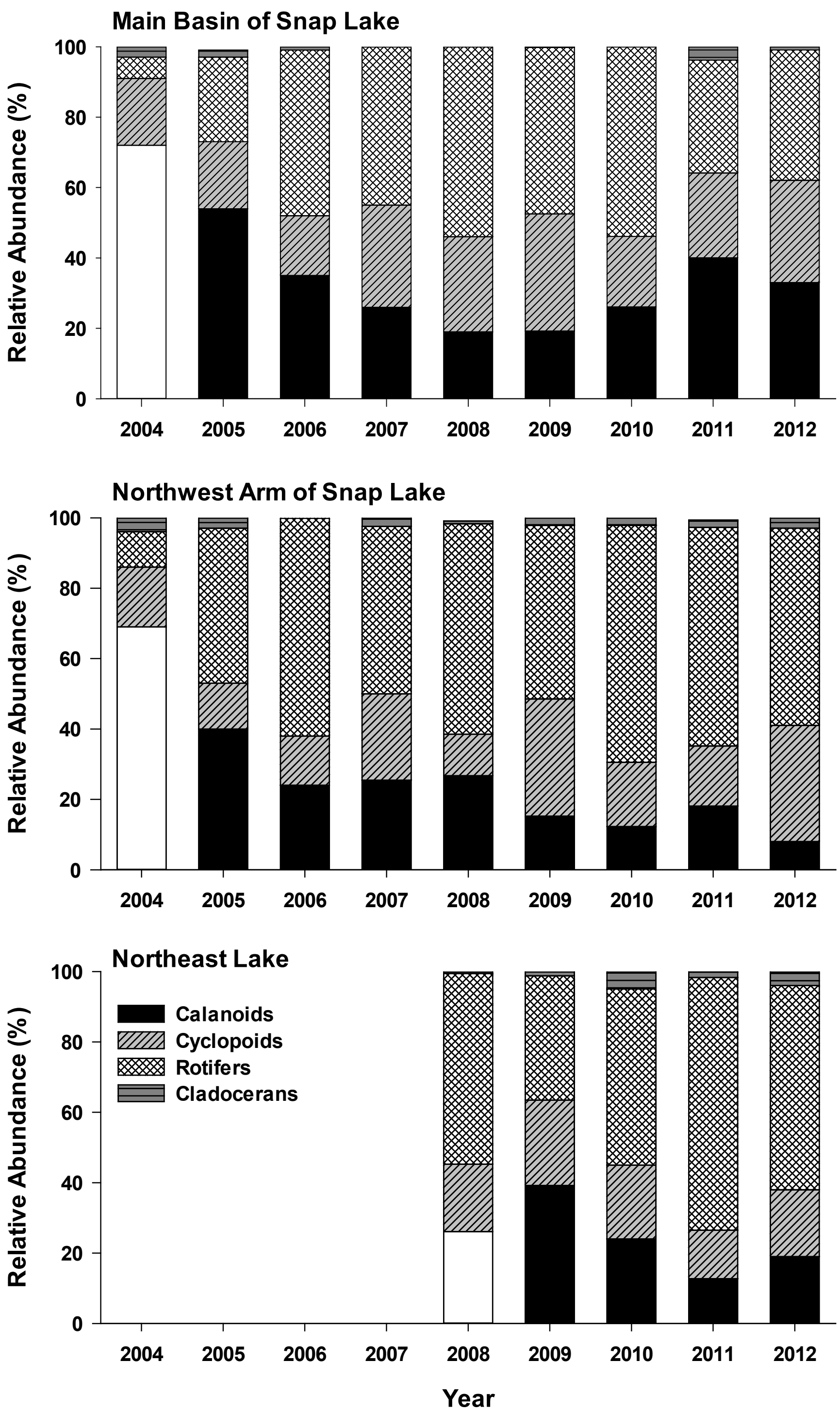
In Northeast Lake, relative zooplankton biomass was dominated by calanoid copepods from 2008 to 2011, with increasing biomass of rotifers. By 2012 the community was co-dominated by calanoid copepods and rotifers, while relative abundance has been consistently dominated by rotifers (Figures 5-27 and 5-28).

Figure 5-27 Relative Biomass of Zooplankton in the Main Basin and Northwest Arm of Snap Lake, 2004 to 2012, and Northeast Lake, 2008 to 2012



Note: Sampling did not occur in Northeast Lake until July 2008 and did not include August sampling until 2011.
% = percent.

Figure 5-28 Relative Abundance of Zooplankton in Snap Lake, 2004 to 2012, and Northeast Lake, 2008 to 2012



Note: Sampling did not occur in Northeast Lake until July 2008 and did not include August sampling until 2011.
% = percent.

5.4.6.3 Zooplankton Community Composition by Species

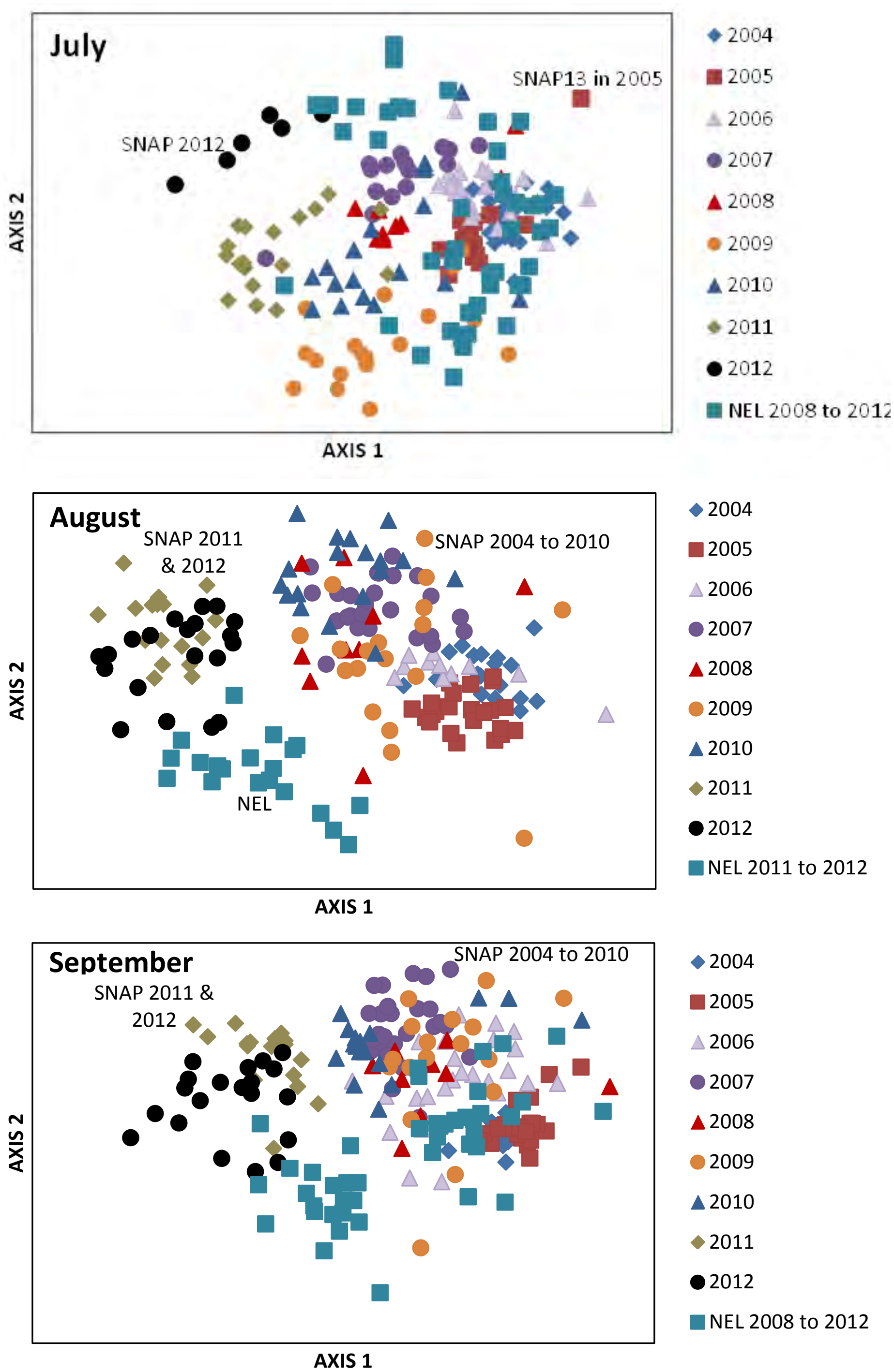
Two-dimensional NMDS configurations showing variation in zooplankton community composition had stress values of 0.22 (July), 0.22 (August), and 0.21(September), indicating a reasonable fit to the original data set.

The July ordination plot showed few patterns in the zooplankton data. The year and station data from Snap Lake clustered with the year and station data from Northeast Lake, with the exception of data from Snap Lake in 2012, which were separate from the main grouping (Figure 5-29).

The August ordination plot showed that the 2004 to 2010 data from Snap Lake grouped separately from the 2011 to 2012 data. In addition, data from Northeast Lake grouped together, but were separate from the Snap Lake data groupings.

In the September ordination plot, patterns of zooplankton community structure in Snap Lake were similar to those in August; the 2004 to 2010 data in Snap Lake grouped together separately from the 2011 and 2012 data. Northeast Lake data from 2008 to 2010 clustered together with Snap Lake data from 2004 to 2010, and Snap Lake data from 2011 and 2012 grouped together with Northeast Lake data from 2011 and 2012.

Figure 5-29 Non-metric Multidimensional Scaling for July, August, and September Zooplankton Biomass in Snap Lake, 2004 to 2012, and Northeast Lake, 2008 to 2012



5.4.7 Reference Lake 13

Results from the single sampling event in August 2012 are summarized below. All comparisons to Snap Lake and Northeast Lake are based on August data only from these two waterbodies. The chlorophyll *a* concentration in Lake 13 (0.68 µg/L) was lower than mean concentrations in Snap Lake (0.99 ± 0.11 µg/L in the main basin and 1.1 ± 0.1 µg/L in the northwest arm), but similar to mean concentrations in Northeast Lake (0.74 ± 0.03 µg/L; Figure 5-30). In contrast, the chlorophyll *c* concentration in Lake 13 (0.06 µg/L) was lower than mean concentrations in Snap Lake (0.09 ± 0.03 µg/L in the main basin and 0.07 ± 0.02 µg/L in the northwest arm) and Northeast Lake (0.11 ± 0.02 µg/L).

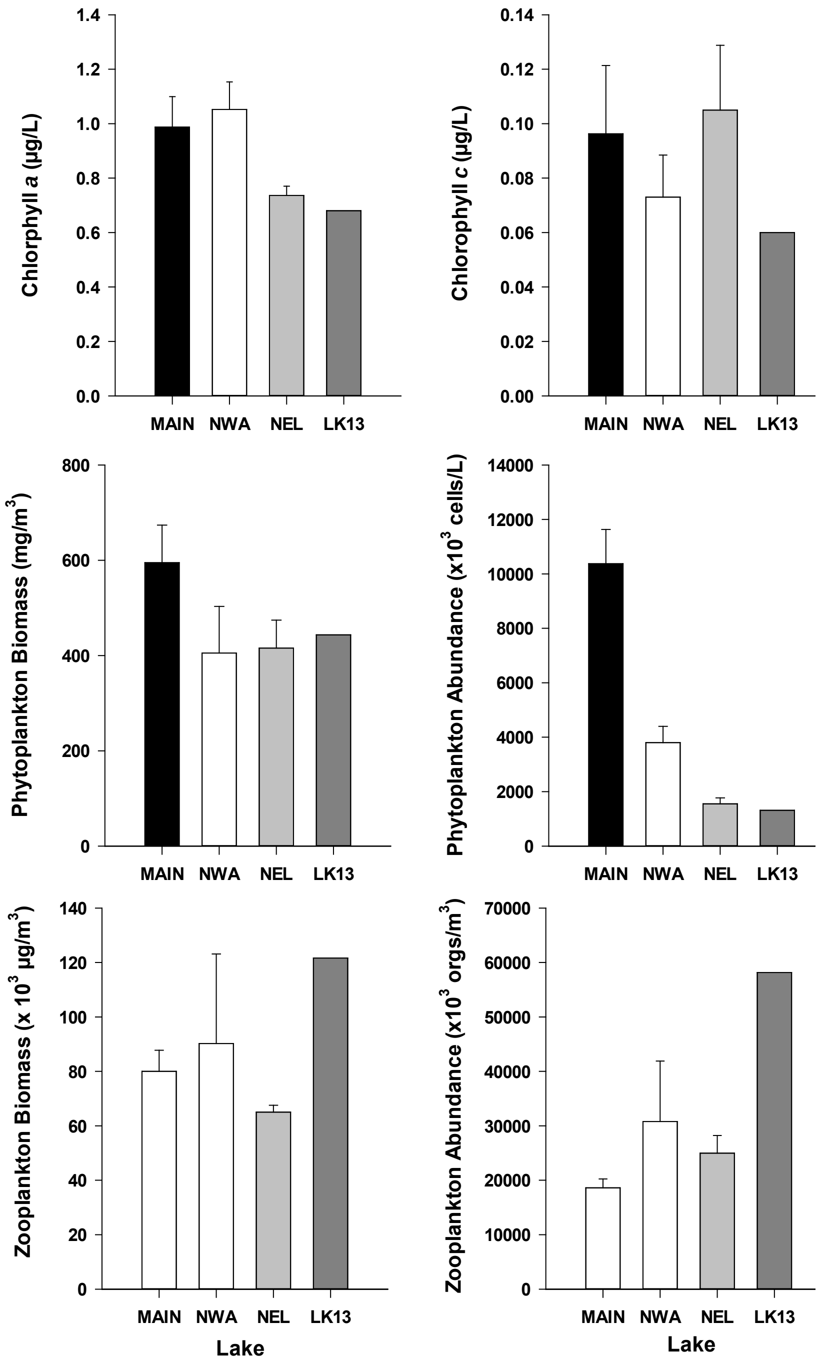
Phytoplankton biomass in the main basin of Snap Lake (594 ± 79 mg/m^3) was greater than observed at the single station sampled in Lake 13 (443 mg/m^3). However, phytoplankton biomass in Lake 13 was slightly greater than in Northeast Lake (415 ± 59 mg/m^3) and the northwest arm of Snap Lake (405 ± 98 mg/m^3; Figure 5-30). Phytoplankton abundance in the main basin of Snap Lake was substantially greater than Lake 13. In addition, phytoplankton abundance in the northwest arm of Snap Lake and Northeast Lake were greater than abundance observed in Lake 13.

The proportion of major taxonomic groups of phytoplankton varied between waterbodies in August. In Lake 13, chrysophytes were the dominant phytoplankton group by biomass, followed by cyanobacteria and chlorophytes (Figure 5-31). Diatom biomass in Lake 13 was minimal (3%) compared to the main basin (68%) and the northwest arm of Snap Lake (26%). Although a higher percentage of chrysophytes (48%) and a lower percentage of cyanobacteria (25%) were observed in Lake 13 compared to Northeast Lake (15% chrysophytes and 70% cyanobacteria), the phytoplankton communities in these two lakes were most comparable. The NMDS results showed that stations in Snap Lake from 2004 and 2005 grouped closely with Northeast Lake (2011 and 2012) and Lake 13 (2012; Figure 5-32).

Zooplankton abundance was higher in Lake 13 (58,147 organisms per cubic metre [orgs/m^3]) compared to the northwest arm of Snap Lake (30,780 ± 11,106 orgs/m^3), the main basin of Snap Lake (18,603 ± 1,638 orgs/m^3), and Northeast Lake (24,985 ± 3,228 orgs/m^3) (Figure 5-30). Similarly, zooplankton biomass was higher in Lake 13 (121,654 µg/m^3) compared to the northwest arm of Snap Lake (90,220 ± 32,900 µg/m^3), the main basin of Snap Lake (80,020 ± 7,728 µg/m^3), and Northeast Lake (65,020 ± 2,537 µg/m^3).

Zooplankton relative biomass in Lake 13 was unlike Snap Lake or Northeast Lake in August. Rotifers dominated the zooplankton community in Lake 13 (76%), while the zooplankton community in the northwest arm of Snap Lake was dominated by cyclopoid copepods (58%); the main basin and Northeast Lake zooplankton communities were dominated by calanoid copepods (49% and 42%, respectively; Figure 5-33). The NMDS ordination plot indicated that the Lake 13 data were different than (i.e., separated from) Snap Lake and Northeast Lake data (Figure 5-34).

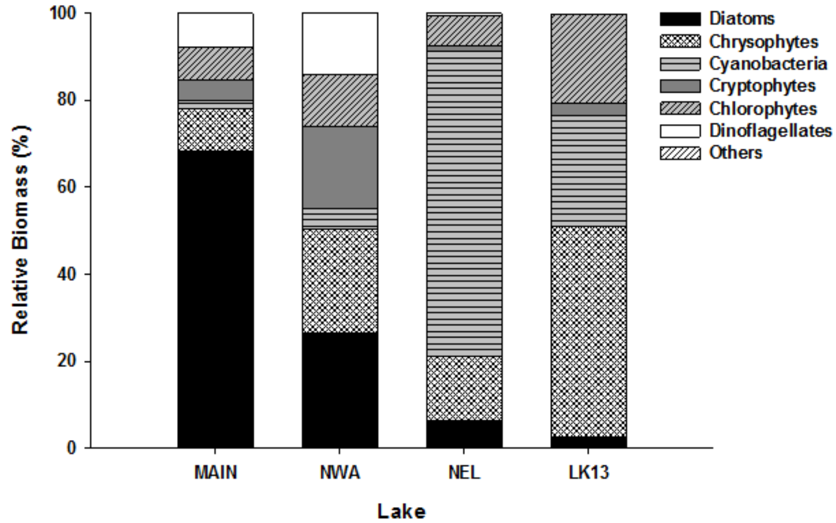
Figure 5-30 Mean Chlorophyll *a*, Chlorophyll *c*, Phytoplankton Biomass, Phytoplankton Abundance, Zooplankton Biomass, and Zooplankton Abundance in the Main Basin and Northwest Arm of Snap Lake, Northeast Lake, and Lake 13, August 2012



Note: Error bars represent standard error of the mean. Sampling occurred at five stations in the main basin of Snap Lake, the northwest arm of Snap Lake, and Northeast Lake, but at only one station in Lake 13.

MAIN= main basin of Snap Lake; NWA= northwest arm of Snap Lake; NEL= Northeast Lake; LK13= Lake 13; µg/L= micrograms per litre; mg/m^3= milligrams per cubic metre; cells/L= cells per litre; orgs/m^3= organisms per cubic metre; µg/m^3= micrograms per cubic metre.

Figure 5-31 Relative Phytoplankton Biomass in the Main Basin and Northwest Arm of Snap Lake, Northeast Lake, and Lake 13, August 2012



Sampling occurred at five stations in the main basin of Snap Lake, the northwest arm of Snap Lake, and Northeast Lake, but at only one station in Lake 13.

MAIN= main basin of Snap Lake; NWA= northwest arm of Snap Lake; NEL= Northeast Lake; LK13= Lake 13;% = percent.

Figure 5-32 Non-metric Multidimensional Scaling for August Phytoplankton Biomass in Snap Lake, 2004 to 2012, Northeast Lake, 2008 to 2012, and Lake 13, 2012

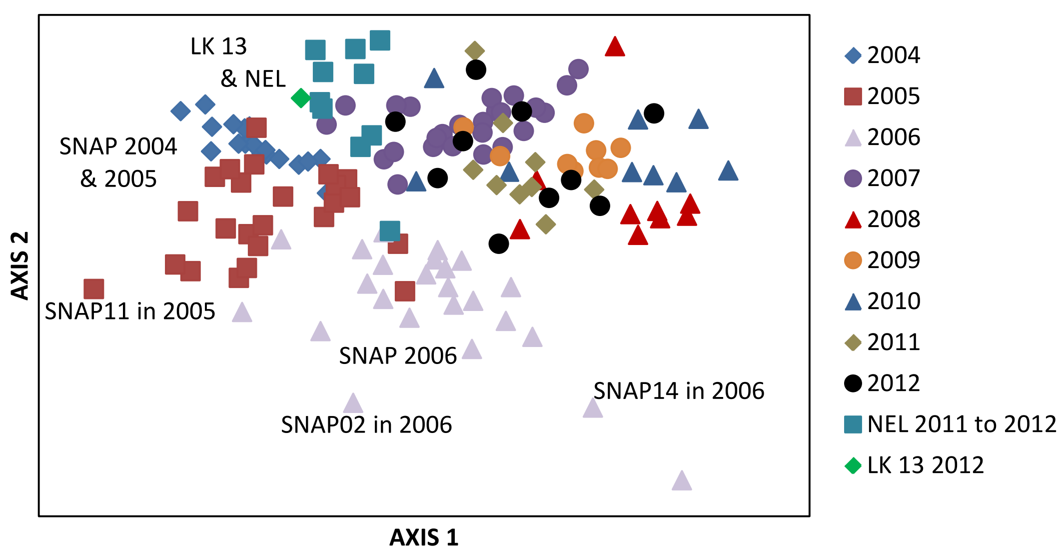
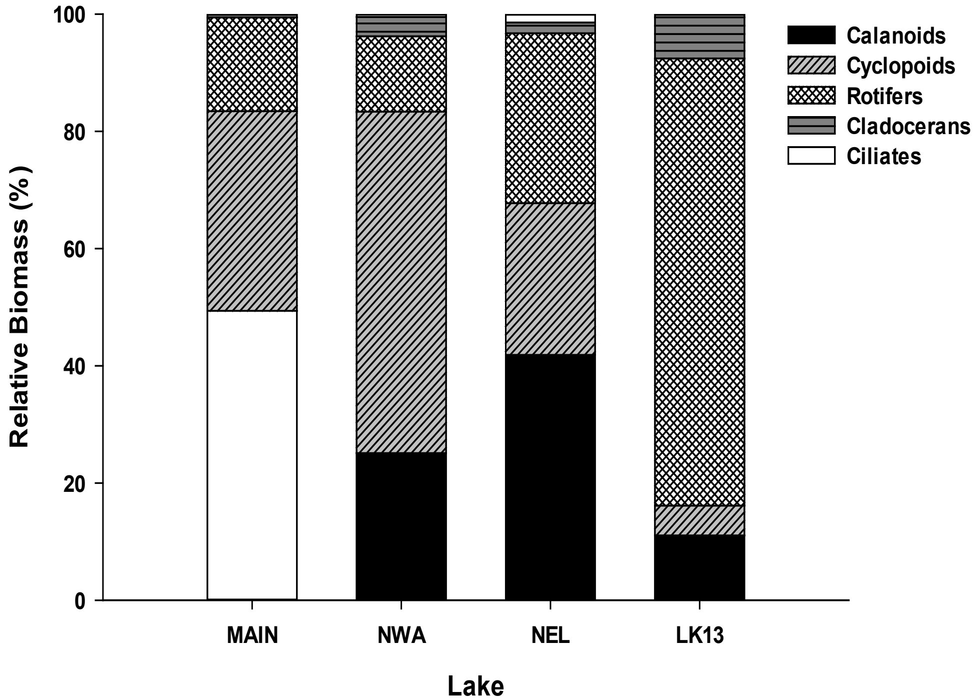


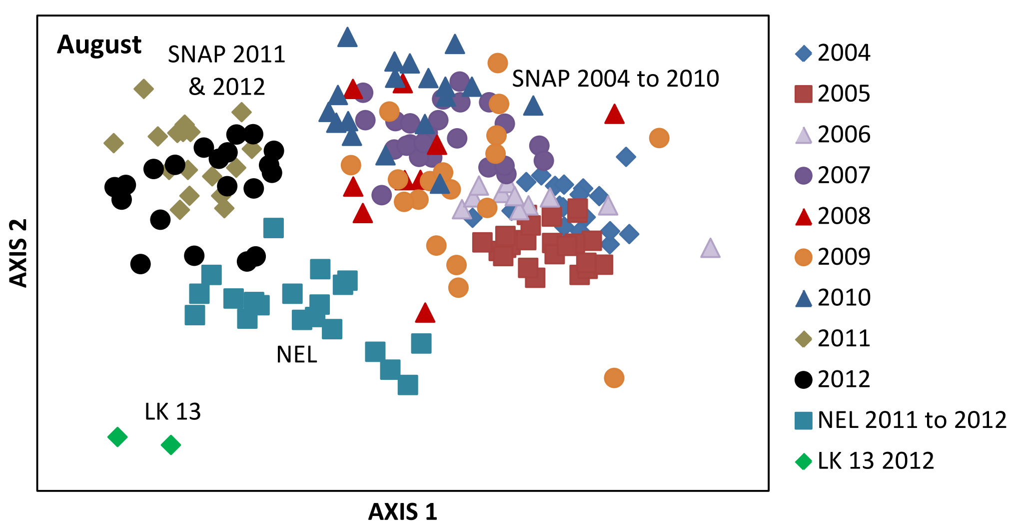
Figure 5-33 Relative Zooplankton Biomass in the Main Basin and Northwest Arm of Snap Lake, Northeast Lake, and Lake 13, August 2012



Sampling occurred at five stations in the main basin of Snap Lake, the northwest arm of Snap Lake, and Northeast Lake, but at only one station in Lake 13.

MAIN= main basin of Snap Lake; NWA= northwest arm of Snap Lake; NEL= Northeast Lake; LK13= Lake 13;% = percent.

Figure 5-34 Non-metric Multidimensional Scaling for August Zooplankton Biomass in Snap Lake, 2004 to 2012, Northeast Lake, 2008 to 2012, and Lake 13, 2012



5.4.8 Picoplankton

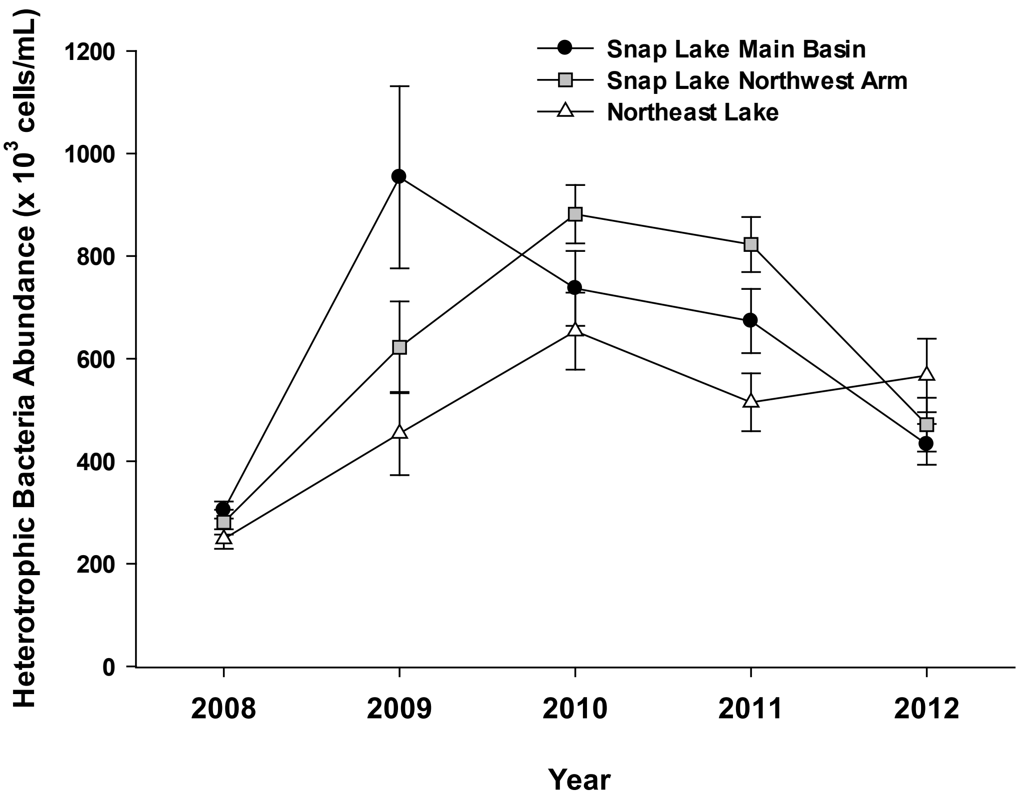
5.4.8.1 Heterotrophic Bacteria

Heterotrophic bacterial abundances in the main basin and northwest arm of Snap Lake were greater than in Northeast Lake from 2008 to 2011 (Figure 5-35). However, since 2009 in the main basin and since 2010 in the northwest arm, a clear decreasing trend has been observed in Snap Lake, which led to lower heterotrophic bacterial abundances between Snap Lake (main basin = 432,886 ± 39,671 cells/mL and northwest arm = 471,103 ± 52,473 cells/mL) and Northeast Lake (567,231 ± 71,791 cells/mL) in 2012.

Mean heterotrophic bacterial abundances in Northeast Lake in July were substantially lower (204,400 ± 10,740 cells/mL) than mean abundances observed in August (701,200 ± 37,800 cells/mL) or September 796,200 ± 36,290 cells /mL; Figure 5-36). The lower abundances in July decreased the annual mean for Northeast Lake, bringing it closer to mean annual abundances observed in Snap Lake (Figure 5-35). In contrast, mean abundances of heterotrophic bacteria in the main basin of Snap Lake in July were greater (296,400 ± 67,600 cells/mL) than those observed in Northeast Lake and were more similar to those observed in the main basin of Snap Lake in August (451,400 ± 39,000 cells/mL) and September (550,800 ± 23,300 cells/mL) (Figure 5-36). Seasonal peaks occurred in either July or September in the northwest arm of Snap Lake (Figure 5-36).

Overall, heterotrophic bacterial abundances increased over the open-water season at most stations in Snap Lake and Northeast Lake, but these trends were more pronounced in Northeast Lake (Figure 5-36). No clear spatial patterns in relation to the diffuser were evident in Snap Lake in 2012 (Figure 5-36).

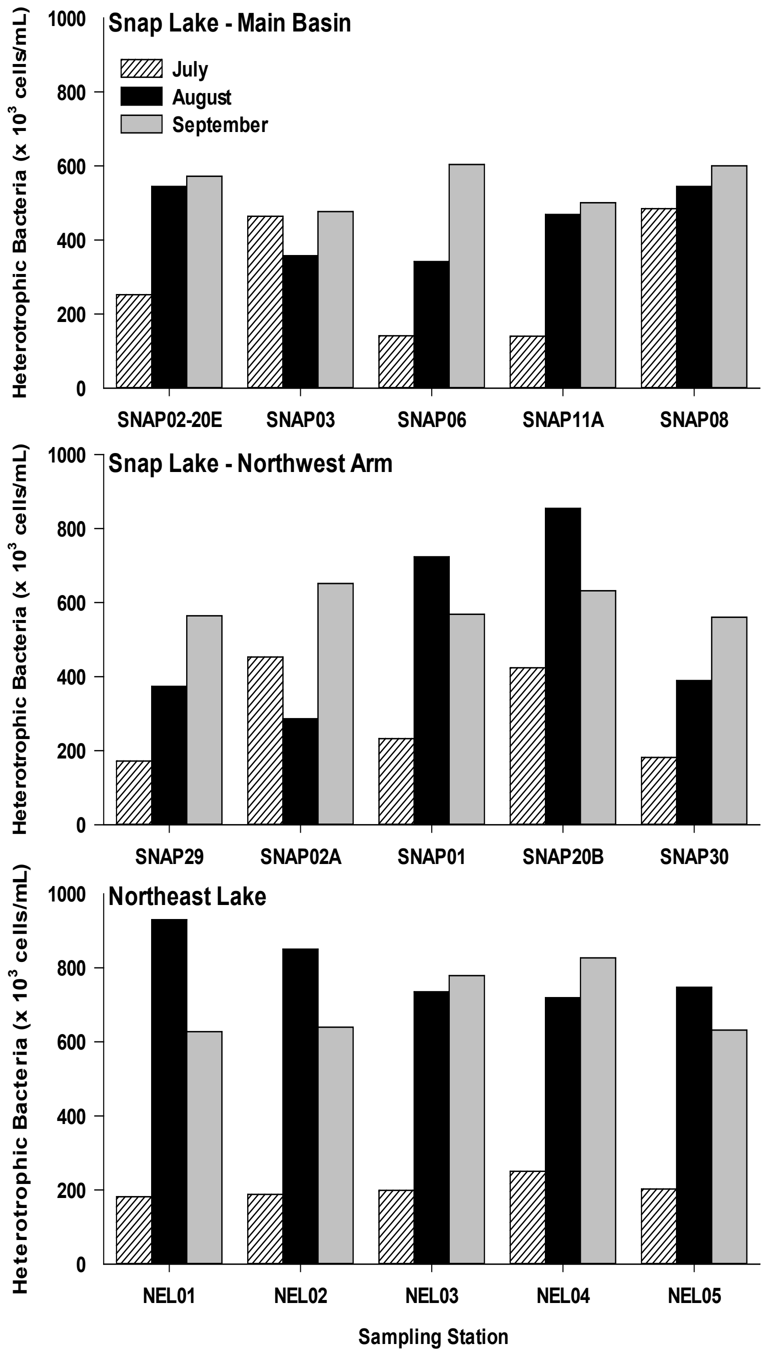
Figure 5-35 Temporal Trends in Mean Annual Heterotrophic Bacterial Abundance in the Main Basin and Northwest Arm of Snap Lake, and Northeast Lake, 2008 to 2012



Note: Error bars represent standard error of the mean. Sampling in Northeast Lake did not include an August sampling session until 2011.

mL = millilitre.

Figure 5-36 Seasonal and Spatial Trends in Heterotrophic Bacterial Abundance in the Main Basin and Northwest Arm of Snap Lake, and Northeast Lake, 2012



Note: Stations are arranged from closest to the diffuser (SNAP29) to farthest from the diffuser (SNAP30) in the main basin and northwest arm of Snap Lake.
mL = millilitre.

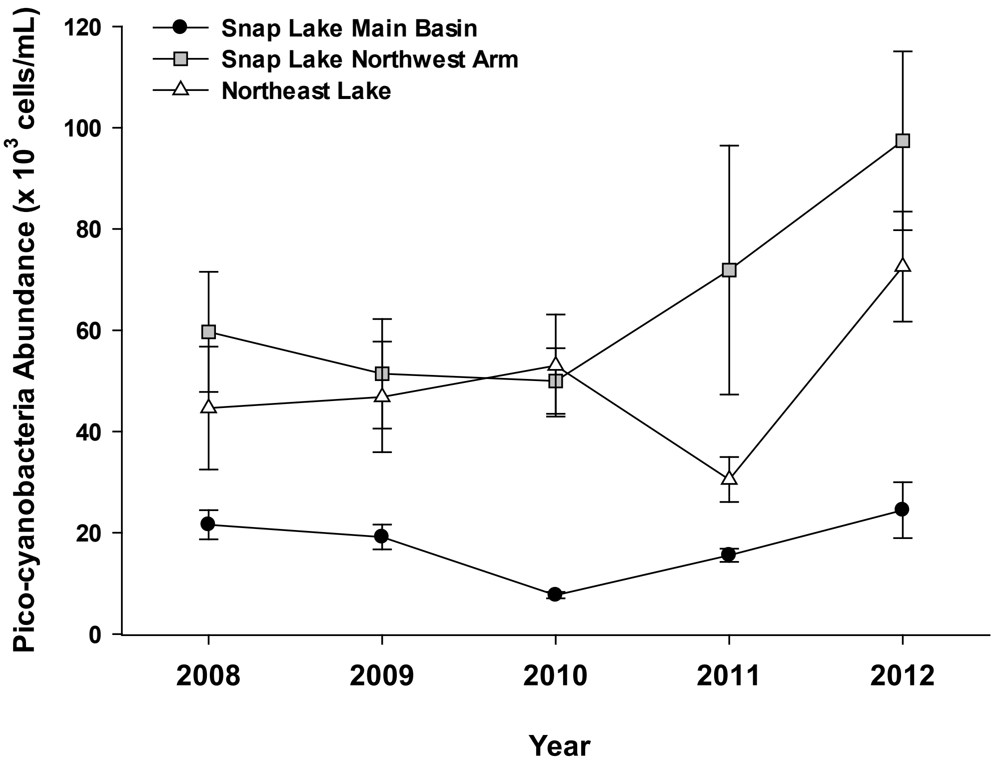
5.4.8.2 Pico-cyanobacteria

Between 2008 and 2012, mean (± SE) annual pico-cyanobacterial abundances were consistently greater in Northeast Lake and the northwest arm of Snap Lake compared to the main basin of Snap Lake (Figure 5-37). Mean (± SE) pico-cyanobacterial abundances in Northeast Lake and the northwest arm of Snap Lake were similar from 2008 to 2010. In 2011, pico-cyanobacterial abundances in Northeast Lake exhibited a decrease (30,511 ± 4,451 cells/mL), while abundances in the northwest arm of Snap Lake increased (71,886 ± 24,588 cells/mL). In 2012, mean pico-cyanobacterial abundances increased in both Northeast Lake and in the northwest arm of Snap Lake. In contrast, pico-cyanobacterial abundances in the main basin of Snap Lake have remained relatively unchanged throughout the monitoring period (15,547 to 24,457 cells/mL), with the exception of 2010, when there was a slight decrease (7,666 cells/mL).

Seasonal peaks in pico-cyanobacterial abundances occurred in September in Northeast Lake and the main basin of Snap Lake in 2012 (Figure 5-38). Seasonal peaks were observed in either August or September in the northwest arm of Snap Lake, depending on the station.

A weak spatial pattern, in relation to the diffuser, was evident in the main basin of Snap Lake in July and September in 2012 (Figure 5-38). Stations closer to the diffuser generally had higher pico-cyanobacterial abundances compared to those further away. There was no spatial pattern evident in the northwest arm of Snap Lake.

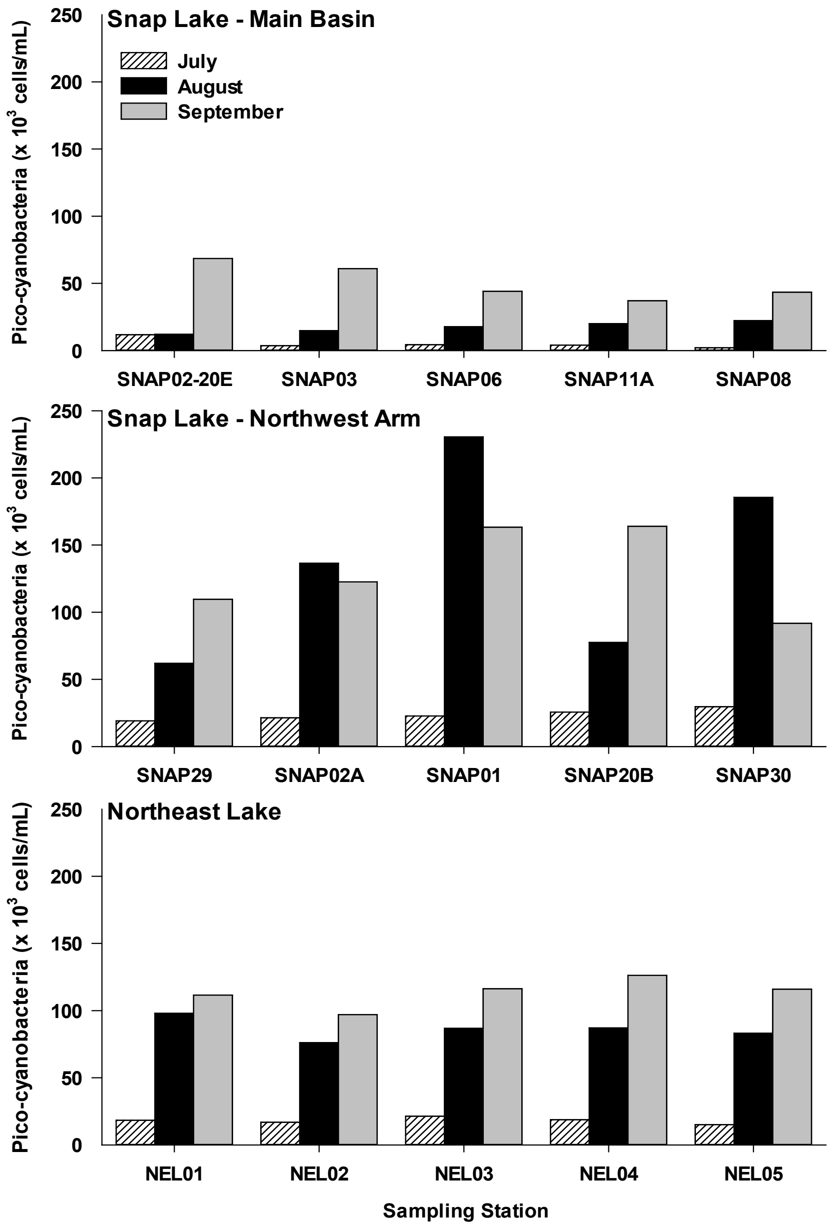
Figure 5-37 Temporal Trends in Mean Annual Pico-cyanobacterial Abundance in the Main Basin and Northwest Arm of Snap Lake, and Northeast Lake, 2008 to 2012



Note: Error bars represent standard error of the mean. Sampling in Northeast Lake did not include an August sampling session until 2011.

mL = millilitre.

Figure 5-38 Seasonal and Spatial Trends in Pico-cyanobacterial Abundance in the Main Basin and Northwest Arm of Snap Lake, and Northeast Lake, 2012



Note: Stations are arranged from closest to the diffuser (SNAP29) to farthest from the diffuser (SNAP30) in the main basin and northwest arm of Snap Lake.

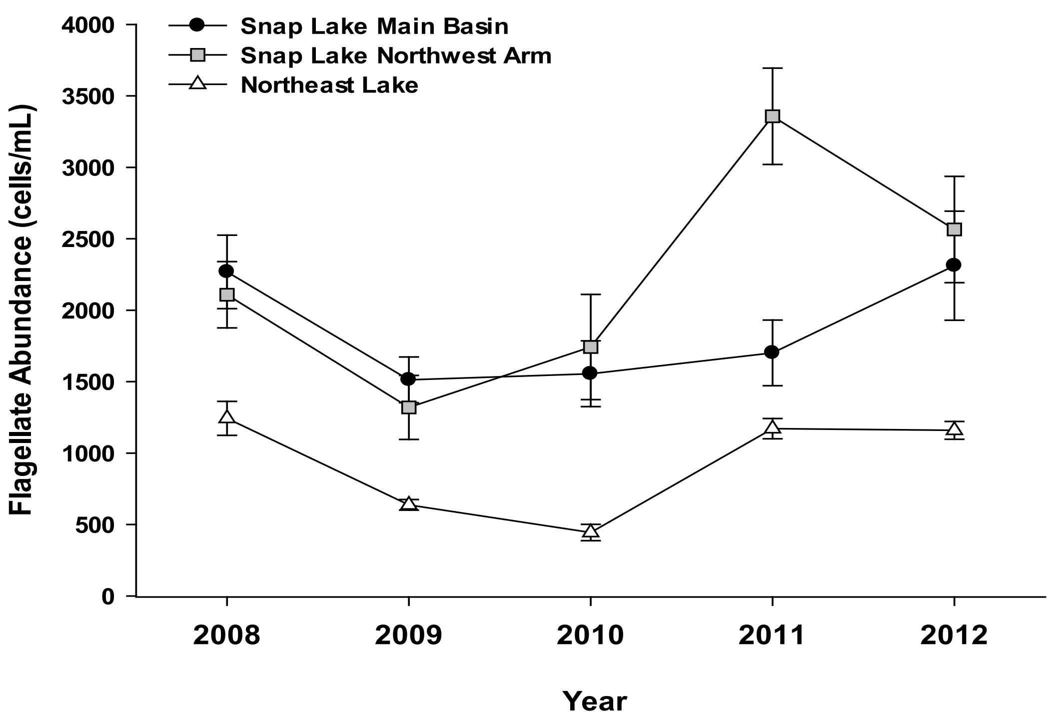
mL = millilitre.

5.4.8.3 Flagellates

The chrysophyte-cryptophyte group dominated the flagellate communities in Snap Lake and Northeast Lake in 2012 (Appendix 5A, Table 5A-8). Overall, mean (±SE) annual flagellate abundance in Northeast Lake (1,159 ± 62 cells/mL) was lower than in Snap Lake (main basin = 2,311 ± 382 cells/mL, and in the northwest arm = 2,565 ± 372 cells/mL; Figure 5-39). The main basin and northwest arm of Snap Lake had similar mean (±SE) flagellate abundances until 2011, when flagellate abundance doubled in the northwest arm (3,357 ± 337 cells/mL) relative to the main basin (1,700 ± 230 cells/mL). In 2012, mean flagellate abundances were similar throughout the lake.

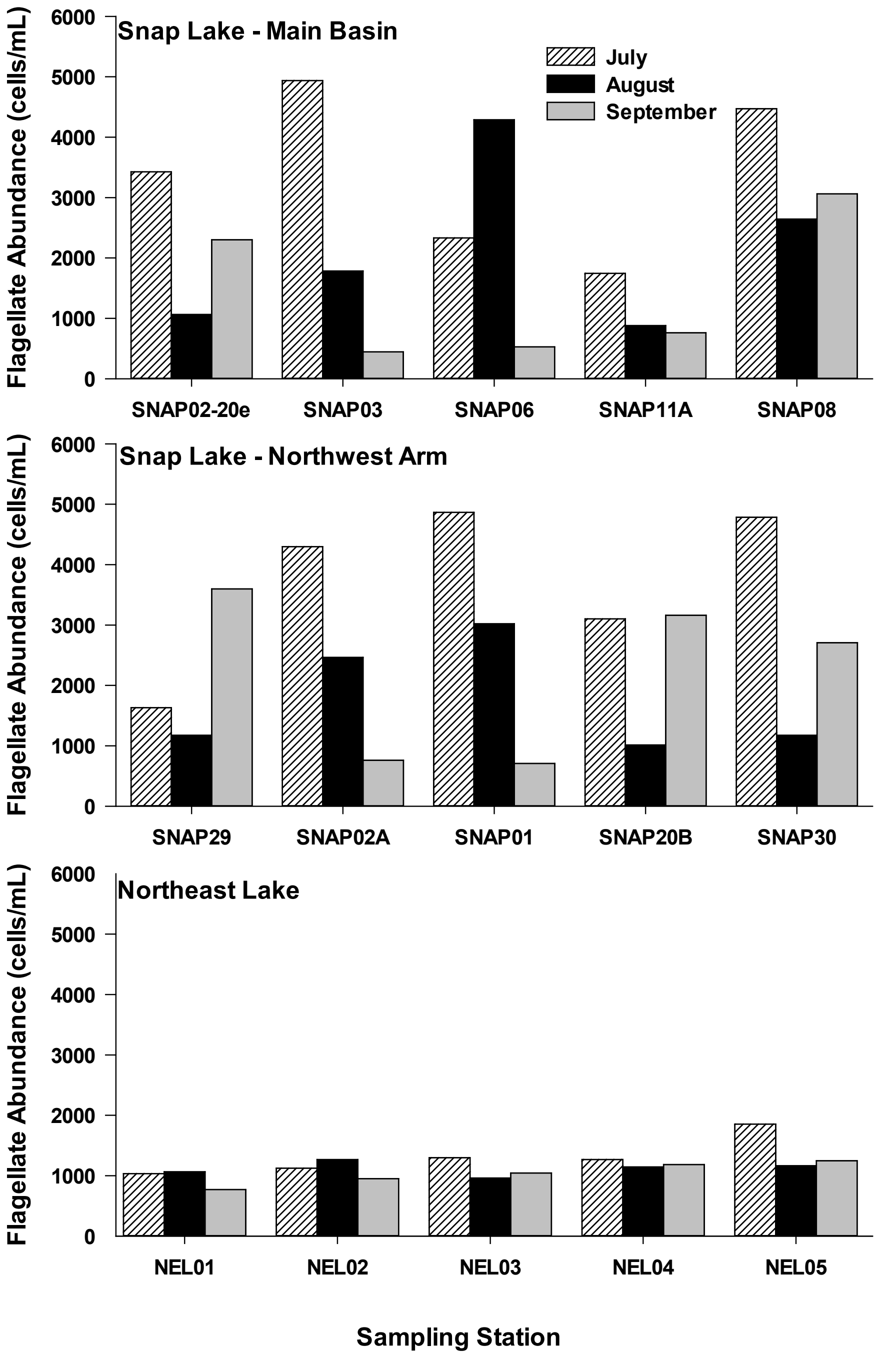
Most stations in Snap Lake and Northeast Lake exhibited peak flagellate abundances in July 2012 (Figure 5-40). In September, there was a strong relationship, with the exception of SNAP02-20e, between proximity to the diffuser and flagellate abundance in the main basin of Snap Lake; stations closer to the diffuser tended to have lower flagellate abundances than those further away (Figure 5-40).

Figure 5-39 Temporal Trends in Mean Annual Flagellate Abundance in the Main Basin and Northwest Arm of Snap Lake, and Northeast Lake, 2008 to 2012



Note: Error bars represent standard error of the mean. Sampling in Northeast Lake did not include an August sampling session until 2011.
mL = millilitre.

Figure 5-40 Spatial and Temporal Flagellate Abundance in the Main Basin and Northwest Arm of Snap Lake, and Northeast Lake, 2008 to 2012



Note: Stations are arranged from closest to the diffuser (SNAP29) to farthest from the diffuser (SNAP30) in the main basin and northwest arm of Snap Lake.

mL = millilitre.

5.5 DISCUSSION

5.5.1 Supporting Environmental Variables

Mean water temperatures were relatively consistent among sampling stations in the main basin of Snap Lake and Northeast Lake in 2012, but were lower and more variable in the northwest arm of Snap Lake. As in previous years, typical seasonal trends were observed, with mean water temperatures peaking in August at most stations in Snap Lake (De Beers 2008, 2009, 2010, 2011, 2012a). A steady decline in water temperature with increasing depth was observed in July at most stations in the main basin of Snap Lake and in Northeast Lake. However, July water profile temperatures in the northwest arm of Snap Lake were more variable in 2012 compared to temperatures observed in 2011 (De Beers 2012a). In September, many stations in Snap Lake had lower mean water temperatures compared to Northeast Lake. This trend was particularly evident in the northwest arm of Snap Lake, where mean water temperatures at all stations were outside the range observed in Northeast Lake.

The water column remained well oxygenated during all sampling events at most stations in Snap Lake and Northeast Lake, and pH values were generally neutral. However, in comparison to 2011 (De Beers 2012a), there were more instances of mean pH values falling below the CWQG pH range for the protection of aquatic life for cold water biota (pH= 6.5 to 9.0; CCME 2007). The pH values slightly below 6.5 occurred at near bottom depths at SNAP08, SNAP02-20e, and SNAP20B. The mean water column pH values for these stations remained above 6.5. The vertical profile patterns observed at SNAP02-20e and SNAP20B in July 2012 were consistent with the 2011 results; the vertical profile pattern observed at SNAP20B has been consistent since 2008 (Section 3).

As in previous years, conductivity in Snap Lake was elevated compared to Northeast Lake, and values were above the baseline level (approximate lake-wide mean of 26 µS/cm; De Beers 2005) in both the main basin and northwest arm of Snap Lake (De Beers 2008, 2009, 2010, 2011, 2012a). Conductivity was elevated at sampling stations in the main basin of Snap Lake compared to sampling stations in the northwest arm of Snap Lake and Northeast Lake. Conductivity in the northwest arm of Snap Lake was higher in the vicinity of the water intake (SNAP29) than at SNAP01 and SNAP30, which are located farther west in the northwest arm of Snap Lake. In the main basin of Snap Lake, values were within the same range at all stations, indicating that the main basin continued to be well mixed in 2012.

Secchi depths in the main basin of Snap Lake and Northeast Lake were similar during the open-water period in 2012, while values were often lower in the northwest arm of Snap Lake, consistent with previous years (De Beers 2012b). From 2008 to 2011, an increasing trend in deeper Secchi depths was observed in the main basin of Snap Lake, indicating increased water clarity in the main basin (De Beers 2012b). However, in 2012 shallower Secchi depths were measured in the main basin of Snap Lake.

5.5.2 Nutrients

The majority of TP concentrations throughout Snap Lake remained within the range characteristic of oligotrophic lakes (TP = 3 to 18 µg/L) as described by Wetzel (2001) and as defined by the CCME (2007; TP = 4 to 10 µg/L) (Appendix 5A, Table 5A-2). The EAR (De Beers 2002) predicted a gradual lake-wide increase in TP from 4 to 12 µg/L to 13 to 23 µg/L over a 20-year period. However, current TP concentrations still fall within the original baseline range and the predicted shift from oligotrophic TP concentrations to the lower to mid-range accepted for mesotrophic lakes has not been observed.

The TN concentrations were above the range for an oligo-mesotrophic lake (307 to 1,387 µg/L; Wetzel 2001) during all three open-water sampling periods in the main basin. The TN concentrations in the main basin of Snap Lake, and at SNAP29 and SNAP02A in the northwest arm, were within the range characteristic of eutrophic lakes (393 to 6,100 µg/L; Wetzel 2001). The three stations further from the diffuser in the northwest arm of Snap Lake (SNAP01, SNAP20B, and SNAP30) had lower TN concentrations, which fell within the range characteristic of mesotrophic lakes (361 to 1,387 µg/L; Wetzel 2001). TN concentrations in Northeast Lake remained within the range characteristic of oligotrophic lakes.

In 2012, the N:P molar ratio indicated that the main basin of Snap Lake was severely P-limited. The N:P molar ratio has increased steadily in the main basin of Snap Lake since 2007 but has remained relatively unchanged in the northwest arm of Snap Lake and Northeast Lake. Based on N:P ratios from 2006, it is assumed that the plankton community in Snap Lake was severely P-limited prior to Mine operation. Mean TP concentrations have remained relatively unchanged within Snap Lake, with concentrations similar to Northeast Lake. Conversely, TN concentrations in the main basin of Snap Lake have exhibited a consistent increasing trend since 2008, while mean concentrations in the northwest arm of Snap Lake and in Northeast Lake have remained relatively unchanged.

In 2012, mean TN concentrations were higher in the main basin of Snap Lake compared to the northwest arm, with stations closer to the diffuser generally having higher concentrations compared to those farther away. The high TN concentrations observed at SNAP29 may be the result of treated effluent mixing into the northwest arm of Snap Lake.

With the increase in TN concentrations in the water column of Snap Lake, a slight increase in P-loading to the system is likely to have little impact on the plankton community. This is because the increasing nitrogen concentrations will continue to shift the community towards a state of P-limitation. However, since Snap Lake is severely P-limited, any P-loaded into the system is likely to be rapidly taken-up and used by the plankton community (Wetzel 2001).

Silicon concentrations were similar between the northwest arm and main basin of Snap Lake, and Northeast Lake in 2005. Since 2005, Si concentrations in the main basin have increased, while a similar trend has not been observed in the northwest arm of Snap Lake or Northeast Lake, where Si concentrations remain close to the 2005 levels. The largest increase in Si concentrations occurred between 2011 and 2012. Although increases in mean Si concentrations were observed in Northeast Lake and the northwest arm of Snap Lake during this same time period, the mean concentrations have remained close to the concentrations measured in 2005. The increased Si concentrations in the main basin of Snap Lake are likely Mine-related, as elevated concentrations are being detected in the treated effluent, but are not being observed in the natural lake water (Section 3).

5.5.3 Chlorophyll *a* and *c*

Chlorophyll *a* and TP are often used as metrics to measure eutrophication (Wetzel 2001), but similar to TP, chlorophyll *a* concentrations in Snap Lake have varied between 2004 and 2012 showing no clear temporal trends in either the main basin or northwest arm of Snap Lake. There is also a lack of a consistent spatial trend in mean chlorophyll *a* concentrations between the main basin and northwest arm of Snap Lake. The EAR (De Beers 2002) predicted that chlorophyll *a* concentrations would gradually increase from a range of 0.2 to 1.8 µg/L to a range of 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 2012, chlorophyll *a* concentrations in the main basin and the northwest arm of Snap Lake remained within the range characteristic of oligotrophic lakes (0.3 to 4.5 µg/L; Wetzel 2001).

It was expected that with the increase in diatom and chrysophyte biomass over cyanobacteria biomass, there would be an increase in concentrations of chlorophyll *c* in 2012 compared to 2005. However, chlorophyll *c* concentrations in Snap Lake in 2005 were similar to those measured in 2012, ranging from 0.01 to 1.7 µg/L.

5.5.4 Microcystin-LR

There was historic concern that the proportion of cyanobacteria in Snap Lake would increase and pose a risk to drinking water quality (De Beers 2005). However, an increase in cyanobacteria does not necessarily result in decreased drinking water quality; therefore, the plankton component monitored both the proportion of cyanobacteria and concentration of the cyanotoxin microcystin-LR, which does pose a threat to drinking water quality at elevated concentrations (De Beers 2012b).

In 2012, most microcystin-LR concentrations in both Snap Lake and Northeast Lake were near or below the detection limit of 0.22 µg/L, which is well below the Health Canada drinking water guideline of 1.5 µg/L (Health Canada 2010). With the exception of 2006, microcystin-LR concentrations in both Snap Lake and Northeast Lake have been consistently near or below the detection limit and below the Health Canada drinking water guideline (De Beers 2012a). Microcystin-LR concentrations in 2006 were measured above detection limits at SNAP12

(0.9 µg/L) and SNAP14 (0.74 µg/L) in July 2006 and August 2006, respectively. The consistently low concentrations of microcystin-LR between 2007 and 2012 suggest that the elevated concentrations observed in Snap Lake in 2006 were related to natural factors (e.g., warmer water temperatures, calmer wind conditions) rather than the discharge of treated effluent.

5.5.5 Phytoplankton

Mean annual phytoplankton biomass at stations in the main basin of Snap Lake exhibited an increasing trend from 2004 to 2009. This trend reversed in 2009 with a decreasing trend from 2009 to 2012. An increasing trend, but with lower biomass than in the main basin, was observed in the northwest arm of Snap Lake between 2004 and 2011, with the exception of decreases in biomass in 2010 and 2012. By 2012, phytoplankton biomass in the northwest arm and main basin of Snap Lake were similar, and also similar to baseline (2004) levels. There has been little change in phytoplankton biomass in Northeast Lake since sampling began in 2008, and biomass values have remained close to Snap Lake baseline levels.

Phytoplankton require light for photosynthesis and growth (i.e., increased biomass). It is possible that changes in light intensity and penetration are responsible for the changes in phytoplankton biomass in Snap Lake since 2004. 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. Water quality is determined by alterations in variables such as TDS, total suspended solids (TSS), and nutrients. Excess N and P can promote the growth of algal blooms, which can decrease light penetration into the water column. High 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 light penetration into the water column (Kirk 1994). Concentrations of TDS and nutrients have increased in Snap Lake since the start of Mine operations and it is possible for changes in light intensity to be partially responsible for the trends and patterns seen in phytoplankton biomass. Given that Secchi disks have higher uncertainty in light measurements than underwater light meters; it is recommended that the latter also be used to measure light levels during future AEMP monitoring.

Chrysophytes and cyanobacteria dominated the plankton assemblage, based on relative biomass from 2004 to 2006 in the main basin of Snap Lake. From 2007 to 2012, the relative proportion of cyanobacteria biomass decreased, and the community shifted to a diatom-chrysophyte co-dominated community. The relative importance of chrysophyte biomass to the phytoplankton assemblage in the main basin of Snap Lake has varied from 2008 to 2012, while diatom biomass has continued to exhibit an increasing trend. The 1999 data collected during the environmental assessment process indicated that diatoms were the dominant taxa (35% to 69%) at the majority of sampling stations throughout the open-water season, with chrysophytes composing 5% to 22% of the phytoplankton community (De Beers 2005). This finding suggests that the community underwent a shift in community composition prior to the commencement of Mine operations.

However, direct comparisons to the 1999 data are limited because of the difference in sampling locations (i.e., near-shore in 1999, open-water in 2004 to 2012). In the northwest arm of Snap Lake, the phytoplankton community composition based on relative biomass remains mainly chrysophyte dominated, with the relative proportion of cyanobacteria and diatoms varying over time.

Relative abundance of major taxonomic groups exhibited similar patterns as observed in the relative biomass within Snap Lake. The main basin of Snap Lake was chrysophyte dominated from 2004 to 2009, with the exception of a substantial increase in chlorophyte relative abundance in 2006. This increase in chlorophyte relative abundance in 2006 was related to an increase in *Choricystis minor* (De Beers 2007), but that species has not been identified in any samples collected since 2006.

Phytoplankton community composition in Northeast Lake differs from that in Snap Lake. Since sampling began in 2008, Northeast Lake has consistently been a cyanobacteria-chrysophyte dominated lake, in terms of both relative biomass and relative abundance. Diatom biomass has been variable and has not displayed a distinct temporal trend. The difference in community structure between Snap Lake and Northeast Lake may be indicative of a natural difference in trophic status or it may be indicative of a Mine-related effect within Snap 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 as 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 effluent, which has caused increased P-limitation. Therefore, the N_2-fixing cyanobacteria do not have a competitive advantage over other groups of algae in the main basin of Snap Lake and cyanobacteria would not be expected to dominate.

In contrast to 2011, the northwest arm of Snap Lake and Northeast Lake did not have similar cyanobacterial biomass (De Beers 2012a). In 2012, cyanobacteria biomass in Northeast Lake was, on average, about ten times higher than in Snap Lake. There was also a large disparity in diatom biomass between the two lakes, with Snap Lake having, on average, four times greater biomass compared to Northeast Lake. This was surprising given that the molar ratios of Northeast Lake and the northwest arm of Snap Lake were similar and much lower than those in the main basin. However, the mean TN concentration in the northwest arm of Snap Lake was higher than in Northeast Lake, and it is possible that there is a large enough concentration of TN in Snap Lake that other environmental factors, such as light intensity, competition, or grazing by herbivores, have become limiting to certain groups of phytoplankton. In addition, increasing concentrations of Si in the main basin could potentially provide diatoms a competitive advantage. In these cases, other algal groups, such as diatoms, could proliferate if silica concentrations are sufficient (>100 µg/L; Reynolds 2006).

Overall, the relative abundance and biomass of diatoms has continued to increase since 2004 in Snap Lake. Diatoms require relatively high concentrations of Si for cell wall development, and Si is often the nutrient limiting planktonic diatom growth in many lakes (Wehr and Sheath 2003). Planktonic species spend most of their life away from concentrated sources of Si; in contrast, littoral diatoms have greater access to Si sources, from both sediment and other substrates. In lakes with long residence times, Si can be depleted by diatom growth and subsequent sinking of frustules to the sediments, resulting in a selective advantage for algal groups that do not require Si for growth. But a sustained water column source of Si can maintain relatively high planktonic diatom biomass and abundance. The increase in diatom biomass in the main basin of Snap Lake may be associated with observed increases in open-water Si concentrations, which can be linked to the Mine. Another possible explanation for the increase in the proportion of planktonic diatoms in Snap Lake is that the diatoms are originating from wave action or algal sloughing in areas of high diatom biomass in the littoral zone (near-shore area; Section 12.1). Centric diatoms, which are planktonic in origin, dominated the diatom assemblage in Snap Lake and Northeast Lake in 2012. However, the remainder of the diatom assemblage in Snap Lake were attached algal species, which are associated with littoral substrates (Wetzel 2001).

Chrysophytes are associated with soft-water lakes that are low in alkalinity (0 to 60 mg/L $CaCO_3$) and conductivity (less than 50 µS/cm). They are also well adapted to low water temperatures and low nutrient concentrations (Wehr and Sheath 2003). Chrysophytes have been consistently dominant in Snap Lake and Northeast Lake since 2004. However, biomass estimates indicate that the relative proportion of chrysophytes has declined over time in Snap Lake. This decrease in chrysophyte biomass in Snap Lake is likely associated with the change in Snap Lake from a soft-water lake to hard-water lake and the substantial increase in conductivity (Section 3), which can both be linked directly to Mine operations.

In addition to N, P, and Si requirements, different phytoplankton groups have different micronutrient requirements, which can affect overall community composition. Micronutrients of interest are calcium (Ca), magnesium (Mg), molybdenum (Mo), and selenium (Se) (Wehr and Sheath 2003). Ca and Mg affect the alkalinity of a waterbody; changes in alkalinity can affect the relative importance of chrysophytes in a community (Wehr and Sheath 2003). Molybdenum is an essential micronutrient for some cyanobacteria as a co-factor for N_2-fixation (Wehr and Sheath 2003). Selenium can affect the abundance of the haptophyte *Chrysochromulina* sp., which requires Se (Wehr and Sheath 2003).

The 2012 phytoplankton community structure, particularly in July, was similar to Northeast Lake and baseline conditions. The July ordination plot showed that stations in Snap Lake in 2004 and stations in Northeast Lake (2008 to 2012) were separated from all of the other years, indicating similarity in community composition between the two lakes during baseline conditions in Snap Lake.

The EAR (De Beers 2002) predicted a slight increase in phytoplankton abundance and biomass and a minor change in phytoplankton community structure, stating that the relative proportion of various species may change, with no loss of species and no major shifts in keystone species. The 2012 results are more supportive of EAR predictions than previous monitoring data which indicated more changes than predicted in the EAR (De Beers 2012b).

5.5.6 Zooplankton

Overall, zooplankton biomass has been variable within Snap Lake. There was an increase in mean annual total zooplankton biomass in the main basin of Snap Lake between 2004 and 2007, followed by a decrease between 2008 and 2009. Zooplankton biomass increased between 2010 and 2011, but decreased again in 2012. Zooplankton biomass in the northwest arm of Snap Lake has been variable, with no evident temporal pattern. In Northeast Lake, a gradual increasing trend was observed from 2008 to 2011, followed by a decrease in 2012. The decrease in zooplankton biomass was observed in both the main basin and Northeast Lake; therefore, it is unlikely that the decrease was Mine-related.

From 2004 to 2012, zooplankton biomass in the main basin of Snap Lake has been consistently dominated by calanoid copepods, with cyclopoid copepods or rotifers as the sub-dominant group. Conversely, zooplankton abundance has shifted from calanoid dominance in 2004 and 2005, to rotifer dominance from 2006 to 2010, to an almost even distribution among the calanoid and cyclopoid copepods and rotifers in 2011 and 2012. In the northwest arm of Snap Lake calanoid copepods have been demonstrating a decreasing trend in dominance since 2004, in both abundance and biomass, while rotifers have been increasing in abundance.

Community composition, by both biomass and abundance in Northeast Lake has been consistent with community composition observed in the northwest arm of Snap Lake. In Northeast Lake, relative zooplankton biomass was dominated by calanoid copepods from 2008 to 2011, with increasing biomass of rotifers. By 2012 the community was co-dominated by calanoid copepods and rotifers, while relative abundance has been consistently dominated by rotifers.

Rotifers have been abundant throughout the monitoring period in Snap Lake, but have accounted for a disproportionately small biomass due to their small size. Compared to the main basin, rotifer biomass and abundance in the northwest arm of Snap Lake and Northeast Lake have always been higher, but these differences have become more pronounced since 2010.

Increased rotifer abundance can be due to nutrient enrichment (Wetzel 2001). However, higher abundances of rotifers have been observed in recent years in both Northeast Lake and the northwest arm of Snap Lake compared to the main basin. Increased rotifer abundance is thus not a useful indicator of Mine-related nutrient enrichment in Snap Lake.

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 main basin of Snap Lake (Section 3). However, from 2004 to 2012, relative percent cladoceran biomass has remained low in all of the lakes.

Greater cladoceran biomass was observed in the available data from Northeast Lake from 2008 to 2012 compared to Snap Lake, and higher cladoceran biomass was observed from 2004 to 2006 in Snap Lake compared to recent years. Between 2007 and 2010, cladoceran biomass was particularly low. However, cladoceran biomass in 2011 was comparable to values between 2004 and 2006. The cause of the temporary decline in cladoceran biomass in Snap Lake is not known. This change in cladoceran biomass was not predicted in the EAR and may not be related to the Mine. Cladoceran biomass in Northeast Lake was also low in 2009, which suggests that a regional phenomenon not associated with the Mine caused the biomass decline in Snap Lake from 2007 to 2010. Cladoceran biomass increased in both the main basin and northwest arm of Snap Lake in 2011, but decreased again in the main basin of Snap Lake in 2012.

In oligotrophic systems, copepods are generally the dominant zooplankton group by abundance (Carney 1990), which is consistent with zooplankton composition in Snap Lake and Northeast Lake. Copepod abundance in Northeast Lake has been consistent with abundances observed in Snap Lake from 2008 to 2012. Although copepod abundances are decreasing in Snap Lake and Northeast Lake, copepod biomass continues to be substantial. The grazing rate of copepods is lower than cladocerans; as a result, copepods do not have as great of an effect on phytoplankton community structure or biomass (Wetzel 2001). In addition, copepods and cladocerans affect nutrient cycling in different ways. Copepods excrete faecal pellets while cladocerans excrete dissolved nitrogen and phosphorus, which regenerates P in soluble available forms. Cladocerans speed nutrient cycling and enhance phytoplankton productivity, which tightens the coupling between phytoplankton and zooplankton. Conversely, increased copepod biomass can lead to a reduced coupling of phytoplankton and zooplankton (Carney 1990).

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) is reducing cladoceran biomass and abundance in Snap Lake and Northeast Lake. However, a clear understanding of this relationship does not currently exist. A stomach contents investigation of Lake Chub occurred in 2012. This investigation showed that Lake Chub stomachs contained mostly Gastropods, Tricoptera, and Chironomidae, but no planktonic species. The stomach contents of other fish species were not investigated. Further assessment of such trophic interactions in Snap Lake is planned (e.g., zooplankton feeding on phytoplankton and in turn being fed on by fish). Specifically, a study examining the stable isotopic signature of prey and predators in Snap Lake is planned for the 2013 field season.

The August and September NDMS ordination plots showed patterns in zooplankton community biomass that indicate a change has occurred in both Snap Lake and Northeast Lake. The 2004 to 2010 data in Snap Lake grouped together separately from the 2011 and 2012 data. The Northeast Lake data from 2008 to 2010 grouped together with Snap Lake data from 2004 to 2010 but were separated from the Snap Lake data from 2011 and 2012. The Northeast Lake data and Snap Lake data from 2011 and 2012 grouped together. This trend of gradual divergence in community structure from the baseline and early years of monitoring in Snap Lake and Northeast Lake is likely reflective of changes in water quality caused by potential regional factors (e.g., temperature and light) rather than Mine-related effects.

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 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. The decrease in zooplankton biomass observed in 2012 is unlikely to be Mine-related, but rather associated with regional factors (i.e., temperature and light) as the decreases in biomass and abundance were observed in both the main basin of Snap Lake and Northeast Lake. The results to date are consistent with the EAR prediction of a minor change in zooplankton community structure.

5.5.7 Reference Lake 13

Plankton reconnaissance in Lake 13 in 2012 was minimal and only included one sampling station in August. Lake 13 was examined for its potential use as second reference lake. Results were assessed to determine Lake 13 similarity to Northeast Lake, which is the current reference lake and is not exposed to treated effluent, and to the northwest arm of Snap Lake, which has been exposed to less treated effluent than the main basin of Snap Lake.

Lake 13 was similar to Northeast Lake and the northwest arm of Snap Lake in some of the plankton parameters monitored but differed in a number of others. Chlorophyll *a* concentrations, phytoplankton abundance, biomass, and community composition were similar between Northeast Lake and Lake 13, but chlorophyll *c*, and zooplankton abundance, biomass, and community composition differed among the three lakes (i.e., Lake 13, Northeast Lake, and the northwest arm of Snap Lake). In particular, zooplankton community composition in Lake 13 was markedly different from Northeast Lake and northwest arm of Snap Lake.

Based on this preliminary evaluation of Lake 13, it may not be an entirely appropriate second reference lake for Snap Lake plankton metrics. However, given the inherent natural variability and patchy distribution of plankton communities, additional samples should be collected before a final decision is made.

5.5.8 Picoplankton

The two major groups of picoplankton are the free living bacteria (heterotrophic bacteria) and the small phytoplankton (pico-cyanobacteria) (Drakare 2002). Flagellated heterotrophic phytoplankton (flagellates) are larger and graze on picoplankton (Hall et al. 1993). Phytoplankton and picoplankton are affected by similar processes. Changes in nutrient content, light availability, grazing, and concentrations of dissolved organic carbon (DOC) affect the interactions of heterotrophic bacteria, pico-cyanobacteria, and large phytoplankton.

Picoplankton are important, and often dominant, primary producers in oligotrophic waters, such as Northeast Lake, as these species are often superior P-competitors over larger taxa due to their small size (Wehr and Sheath 2003). A small size is advantageous when nutrients are present in low concentrations, because small cells have a large surface area to volume-ratio, which enhances metabolic rates relative to larger cells (Drakare 2002). In lakes, picoplankton favour good light conditions, and typically picoplankton abundance will be higher in clearer water.

Growth rates of autotrophic picoplankton in ultra-oligotrophic and meso-oligotrophic lakes have been shown to be inhibited by additions of P and N (Schallenberg and Burns 2001; Stockner and Shortreed 1994). When nutrients are in excess, larger phytoplankton have a competitive advantage over picoplankton, as they are able to quickly assimilate the available nutrients and increase in numbers to the point where they are able to limit light availability to the picoplankton (Drakare 2002).

Heterotrophic bacteria require DOC as an energy source, while phytoplankton and pico-cyanobacteria require solar energy. The presence of a suitable energy source could be more important for the outcome of competition than nutrient uptake ability. Therefore, in principle, pico-cyanobacteria and heterotrophic bacteria should compete on equal terms when both are limited more by nutrients than by the supply of energy (Drakare 2002). In a clear water lake, pico-cyanobacteria can be expected to have greater productive importance relative to heterotrophic bacteria (Drakare 2002). Both pico-cyanobacteria and heterotrophic bacteria are superior competitors compared to larger phytoplankton for nutrients at low concentrations because of their small size. However, heterotrophic bacteria are heavily dependent on DOC produced by phytoplankton in clear water lakes; in lakes with inflow of humic DOC from the drainage area heterotrophic bacteria have an alternative energy source. But when heterotrophic bacteria have only phytoplankton dependent DOC, their dissolved phosphorus uptake rates are similar to phytoplankton (Drakare 2002). It is possible that the decrease observed in heterotrophic bacterial abundance in Snap Lake is linked to the decrease in phytoplankton biomass and thus decreased phytoplankton derived-DOC.

Within Snap Lake, a weak spatial pattern in relation to the diffuser was observed in July and September 2012. Stations closer to the diffuser generally had higher pico-cyanobacterial abundances compared to those further away. This finding was in contrast to the expected trend of pico-cyanobacterial inhibition closer to the diffuser, as growth rates are often inhibited by additions of N and P (Stockner and Shortreed 1994; Schallenberg and Burns 2001). A spatial pattern was also observed in flagellate abundances in Snap Lake in September. With the exception of SNAP02-20e, there was a strong relationship between proximity to the diffuser and flagellate abundance in the main basin of Snap Lake; stations closer to the diffuser tended to have lower flagellate abundances than those further away. This finding is in contrast to what was observed in previous years in which there was no discernible pattern in flagellate abundance in relation to proximity to the diffuser in Snap Lake.

A link between pico-cyanobacterial abundance and flagellate abundance was observed in Snap Lake and Northeast Lake. Overall, flagellate abundance was lower in Northeast Lake compared to Snap Lake, while pico-cyanobacteria abundances were higher in Northeast Lake compared to Snap Lake. In addition, seasonal peaks in flagellate abundance occurred in July, whereas pico-cyanobacterial abundance peaks occurred in September at most stations in Northeast Lake and the main basin of Snap Lake, suggesting that flagellate grazing may be influencing abundances in the pico-cyanobacteria community.

It possible that pico-cyanobacteria in Snap Lake are exhibiting a nutrient enrichment response. The lower pico-cyanobacterial abundance in the main basin of Snap Lake and the highly negative correlation to TN (De Beers 2012b) may be indicators of Mine-related N-enrichment. However, as previously noted, this same response was not found close to the diffuser, which argues against a nutrient enrichment response.

5.6 CONCLUSIONS

5.6.1 What are the Current Concentrations of Chlorophyll *a* and *c*, and What Do These Concentrations Indicate About the Trophic Status of Snap Lake and Northeast Lake?

Chlorophyll *a* concentrations in Snap Lake have varied between 2004 and 2012 and no clear temporal trend in either the main basin or northwest arm has been apparent. Chlorophyll *c* concentrations have not increased in Snap Lake since sampling began in 2005. There have been no consistent spatial trends in mean chlorophyll a and c concentrations between the main basin and northwest arm of Snap Lake. In 2012, chlorophyll *a* concentrations in the main basin and the northwest arm of Snap Lake remained within the range characteristic of oligotrophic lakes (0.3 to 4.5 µg/L; Wetzel 2001).

5.6.2 What is the Current Status, in Terms of Abundance, Biomass and Composition, of the Phytoplankton Community in Snap Lake and Northeast Lake, and do these Results Suggest Signs of Mine-Related Nutrient Enrichment or Toxicological Impairment?

In 2012, mean annual phytoplankton biomass within the main basin of Snap Lake was approximately 1.7 times higher than in 2004. Phytoplankton biomass in the northwest arm and main basin of Snap Lake were similar in 2012, with both areas of Snap Lake returning to near-baseline (2004) levels after increasing and decreasing trends from 2004 to 2011. Northeast Lake has not changed substantially since 2008.

In 2012, the main basin of Snap Lake continues to be a diatom-chrysophyte co-dominated community. The relative contribution of chrysophyte biomass to the phytoplankton assemblage has varied from 2008 to 2012, while diatom biomass exhibited an increasing trend from 2004 to 2012. In the northwest arm of Snap Lake, phytoplankton community composition has been mainly chrysophyte dominated, with the relative proportion of cyanobacteria and diatoms varying 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.

The observed increases in diatom biomass since 2004 in the main basin and northwest arm of Snap Lake may be associated with the observed increases in open-water Si concentrations which can be linked to the Mine. Similarly, the decrease in chrysophyte biomass in the main basin of Snap Lake may be associated with the change in Snap Lake from a soft-water lake to hard water lake and the substantial increase in conductivity, which can both be linked directly to Mine operations. However, the 2012 results suggest that current community structure, particularly in July, is more similar to Northeast Lake and baseline conditions, than previous years in Snap Lake. Thus, to date major changes in the phytoplankton community have not occurred.

5.6.3 What is the Current Status, in Terms of Abundance, Biomass and Composition, of the Zooplankton Community in Snap Lake and Northeast Lake, and do these Results Suggest Signs of Mine-Related Nutrient Enrichment or Toxicological Impairment?

Mean annual zooplankton biomass in the main basin of Snap Lake was comparable to Northeast Lake in 2012. However, biomass in the northwest arm of Snap Lake was nearly twice as high as in Northeast Lake and the main basin of Snap Lake. Overall, zooplankton biomass was highly variable, showing no clear trends in the main basin of Snap Lake from 2004 to 2012. Between 2004 and 2007, an increase in mean annual total zooplankton biomass was observed in the main basin of Snap Lake, followed by a decrease between 2008 and 2009. Zooplankton biomass increased between 2010 and 2011, but decreased by approximately 50% between 2011 and 2012. Similarly, zooplankton biomass in the northwest arm of Snap Lake has been variable with no clear trends. In Northeast Lake, a slight increasing trend was observed between 2008 and 2011, followed by a decrease in 2012.

From 2004 to 2012, relative zooplankton biomass in the main basin of Snap Lake has been consistently dominated by calanoid copepods, with cyclopoid copepods or rotifers as the sub-dominant group. Zooplankton abundance has shifted dominance amongst the calanoids, cyclopoids, and rotifers. In contrast, in the northwest arm of Snap Lake calanoid copepods have shown a decrease in dominance, abundance, and biomass since 2004 with an increase in the relative proportion of rotifer and cyclopoids. In Northeast Lake, relative zooplankton biomass has been consistently dominated by calanoid copepods, while relative abundance has been consistently dominated by rotifers.

Greater cladoceran biomass was observed in Northeast Lake from 2008 to 2012 compared to Snap Lake. Higher cladoceran biomass was observed from 2004 to 2006 in Snap Lake compared to recent years. Between 2007 and 2010 and in 2012 cladoceran biomass was particularly low in the main basin of Snap Lake.

The decrease in zooplankton biomass observed in 2012 is unlikely to be Mine-related but rather associated with regional factors (e.g., temperature and light) as the decrease in biomass and abundance was observed in both the main basin of Snap Lake and Northeast Lake. The change in cladoceran biomass is contrary to what was predicted in the EAR. Cladoceran biomass in Northeast Lake was also low in 2009 and in 2012, which suggests that a regional phenomenon not associated with the Mine caused the decline in Snap Lake from 2007 to 2010 and in 2012.

Patterns in zooplankton community biomass indicate that a change has occurred in both Snap Lake and Northeast Lake. This trend of gradual divergence in community structure from the baseline and early years of monitoring in both Snap Lake and Northeast Lake is likely reflective of changes in water quality caused by potential regional factors (e.g., temperature and light) rather than Mine-related effects.

5.6.4 How do Observed Changes Compare to Applicable Predictions in the EAR?

The EAR predicted a gradual lake-wide increase in TP from 4 to 12 µg/L to 13 to 23 µg/L over a 20-year period. However, current TP concentrations still fall within the original baseline range and the predicted shift from oligotrophic TP concentrations to the lower to mid-range accepted for mesotrophic lakes has not been observed.

The EAR also predicted that chlorophyll *a* concentrations would gradually increase from 0.2 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 2012, chlorophyll *a* concentrations in the main basin and the northwest arm of Snap Lake remained within the range characteristic of oligotrophic lakes (0.3 to 4.5 µg/L; Wetzel 2001).

Changes in mean total phytoplankton biomass and abundance between 2004 and 2012 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; the plankton community in Snap Lake in 2012 was similar to baseline conditions. The 2012 phytoplankton data are consistent with the EAR prediction of a minor change in phytoplankton community structure including a change in the relative proportion of various species. However, the EAR also predicted that no loss of species or major shifts in keystone species would occur. The 2012 results are more supportive of EAR predictions than previous monitoring data, which indicated more changes than predicted in the EAR (De Beers 2012b). However, shifts in species composition have been documented, particularly in the diatoms, which now show a greater proportion being of benthic rather than planktonic origin.

The zooplankton community results indicate that low magnitude changes have occurred since baseline conditions (2004) in Snap Lake consistent with the EAR, which 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. The decrease in zooplankton biomass observed in 2012 is unlikely to be Mine-related but rather associated with regional factors as the decreases in biomass and abundance were observed in both the main basin of Snap Lake and Northeast Lake. The decrease in cladoceran biomass in Snap Lake and Northeast Lake is also unlikely to be Mine-related and is contrary to the increase in cladoceran biomass with increases in calcium that was predicted in the EAR.

5.6.5 How does the Plankton Community in Reference Lake 13 Compare to Snap Lake and Northeast Lake? Is Reference Lake 13 a Suitable Reference Lake for Plankton?

Based on the current preliminary evaluation of Lake 13, it may not be an entirely appropriate second reference lake for Snap Lake plankton metrics. However, given the inherent natural variability and patchy distribution of plankton communities, additional samples should be collected before a final decision is made.

5.6.6 What is the Current Status, in Terms of Abundance, of the Picoplankton Community in Snap Lake and Northeast Lake, and do these Results Provide any Evidence of Mine-Related Nutrient Enrichment?

From 2008 to 2011 heterotrophic bacteria abundances in the main basin and northwest arm of Snap Lake were greater than in Northeast Lake. However, decreasing trends in the main basin and northwest of Snap Lake led to similar heterotrophic bacterial abundances between Snap Lake and Northeast Lake in 2012. Conversely, between 2008 and 2012, mean annual pico-cyanobacterial abundances have been consistently greater in Northeast Lake and the northwest arm of Snap Lake compared to the main basin of Snap Lake.

Mean pico-cyanobacterial abundances in Northeast Lake and the northwest arm of Snap Lake were similar from 2008 to 2010. In 2011, pico-cyanobacterial abundances in Northeast Lake exhibited a decrease, while abundances in the northwest arm increased. In 2012, mean pico-cyanobacterial abundances increased in both Northeast Lake and in the northwest arm of Snap Lake. In contrast, pico-cyanobacterial abundances in the main basin of Snap Lake have remained relatively unchanged. The utility of lower pico-cyanobacterial abundance as an indicator of Mine-related N-enrichment is uncertain as lower abundance was found in the main basin of Snap Lake but not close to the diffuser. Pico-cyanobacterial abundance may be affected by other factors than the Mine.

5.6.7 How do the Observed Changes in the Picoplankton Community Compare to Changes Observed in the Phytoplankton Community?

Prior to 2012, phytoplankton and picoplankton results supported a nutrient enrichment hypothesis. The 2012 results suggest other changes in water quality (e.g., increased TDS and alkalinity) may also be influencing the phytoplankton community. As with phytoplankton, the 2012 picoplankton results suggest additional factors may be applicable.

Heterotrophic bacteria mean annual abundance decreased within Snap Lake and no spatial trends in relation to the diffuser were evident. Mean annual total phytoplankton biomass also decreased in Snap Lake, which may be Mine-related (e.g., alteration in TDS and TSS). It is possible that the decrease in heterotrophic bacteria is related to a decrease in phytoplankton-derived DOC.

There appears to be a link between pico-cyanobacteria abundance and flagellate abundance in 2012. It is possible that pico-cyanobacteria in Snap Lake, particularly in the main basin, are exhibiting a nutrient enrichment response. However, the lack of spatial trends in relation to the diffuser contradicts the nutrient enrichment hypothesis. Differences in seasonal peaks in pico-cyanobacteria abundance and flagellate abundance suggest that flagellate grazing may be influencing the pico-cyanobacteria.

5.7 RECOMMENDATIONS

Based on the results to date, the following recommendations are provided for the plankton program:

- Increased Si concentrations may be allowing for greater growth of diatoms. Data on Si should be collected as part of the plankton component of the AEMP rather than solely as part of the water quality component of the AEMP. This would provide depth-integrated samples that could be directly compared to samples collected during the plankton program, and would result in a better understanding of the quantity of Si that is available to diatoms in both lakes.
- An underwater light meter should be used in addition to Secchi depth to measure light penetration into the water column. Light penetration may be a major variable affecting plankton that needs to be measured with less uncertainty than with a Secchi disk.
- Additional evaluation should be undertaken of the suitability, in terms of plankton metrics, of Lake 13 as an appropriate second reference lake.

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6 BENTHIC INVERTEBRATE COMMUNITY

6.1 INTRODUCTION

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. They 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 large 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 Snap Lake and Northeast Lake in September 2012. It also includes data collected in August 2012 from Lake 13, a provisional second reference lake. Benthic invertebrate community characteristics were summarized and benthic community variables were compared statistically among these lakes. Snap Lake, which is exposed to treated effluent from the Snap Lake Mine (Mine), was compared to both Northeast Lake and Lake 13, used as reference lakes. In addition, benthic community variables were compared over time using 2009 to 2012 data collected during the fall.

The 2012 benthic invertebrate program represents the eighth year of benthos monitoring in Snap Lake under the Aquatic Effects Monitoring Program (AEMP) (De Beers 2005a). 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 was used to design the current AEMP (De Beers 2005a). Changes were made after analysis of the 2005 benthic invertebrate data indicated that water depth was a confounding factor that interfered with the detection of potential Mine-related effects.

The 2012 AEMP benthic invertebrate community program represents the fourth year when the full benthic invertebrate program was conducted during the fall open-water season. 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 was not observed during winter. Therefore, the reason for conducting the benthic invertebrate program under-ice, which was to assess the benthos under low oxygen conditions, was found to be not applicable.

The 2012 survey was a control/impact sampling design. During the fall program, three to five stations were sampled in each of the designated sampling areas, including five stations in Northeast Lake, Lake 13, and in each of the former near-field and mid-field areas of Snap Lake. The near-field exposure area was located in the north basin of Snap Lake and the mid-field exposure area was located in the south basin. Beginning in 2012, the near-field and mid-field areas have been combined into a single area referred to as the main basin for comparisons because exposure to treated effluent has been similar for the last five years. Three stations were also sampled in the northwest arm of Snap Lake, which was variably exposed to treated effluent. Exposure to treated effluent was estimated based on conductivity.

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 MV2011L2-0004 (Water Licence) (MVLWB 2012) 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 Environmental Assessment predictions relating to the trophic and DO status of Snap Lake.

The objective of the 2012 Snap Lake benthic invertebrate community survey was to address the following two key questions:

1. In 2012, was the benthic invertebrate community affected by the changes in water and sediment quality in Snap Lake?; and,
2. If the benthic invertebrate community was affected, was the change greater than that stated in the EAR?

Potential effects on the benthic community are related to the following changes in water and sediment quality predicted in the EAR (De Beers 2002):

- A gradual lake-wide increase was predicted in total dissolved solids (TDS), expressed as elevated concentrations in deep parts of the lake under ice-covered conditions, and throughout the water column during the open-water season. Whole-lake mean TDS concentration was predicted to increase over time from a baseline mean of about 15 milligrams per litre (mg/L) to a maximum of 330 mg/L by year 19 from the start of construction. The predicted magnitude of this effect on benthic invertebrates was classified as low, with potential effects primarily due to predicted increases in calcium concentration.
- An increase was predicted in nitrate concentration from a baseline mean of 0.024 mg/L to a maximum whole-lake mean of 5.28 mg/L by year 19, accompanied by a decrease in total phosphorus concentration from 0.010 to 0.005 mg/L. These changes in nitrate concentrations were expected to result in an initial tripling of chlorophyll *a* concentration during construction, followed by a sharp decline close to the baseline level of 0.9 micrograms per litre (µg/L). A gradual increase was predicted through operations to 1.3 µg/L by year 19. The increase in chlorophyll *a* concentration could affect the benthic community via increased food availability. The predicted magnitude of this effect on benthic invertebrates was classified as low.
- Reduced DO concentrations were predicted during winter in deep areas of Snap Lake, from nitrification of ammonia, and breakdown of labile organic matter. The affected area was predicted to be less than 10 percent (%) of the surface area and the bottom area of Snap Lake, with DO concentrations above the threshold for sensitive benthic invertebrates predicted to decrease from 98% to 96%. The predicted magnitude of this effect on water quality was predicted to be negligible; therefore, biological effects were not classified.
- Slight increases in the concentration of hexavalent chromium were predicted in the mixing zone and potentially in bottom sediments; the predicted magnitude of this effect on benthic invertebrates was classified as negligible.

6.2 METHODS

6.2.1 Field Survey

6.2.1.1 Definitions of Terms

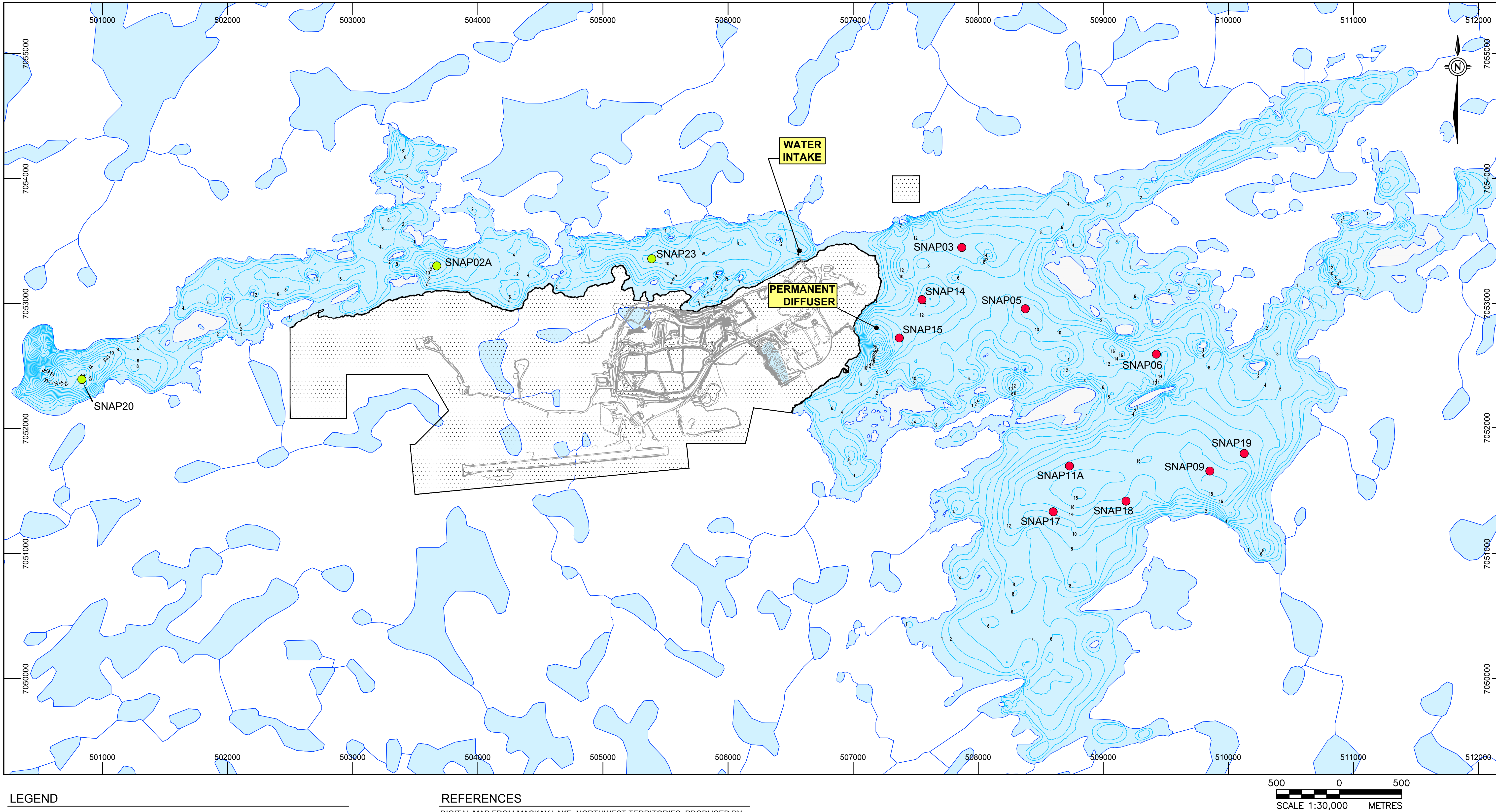
The AEMP was designed according to study design principles of metal mining aquatic environmental effects monitoring (EEM), as described in the metal mining guidance document (Environment Canada 2012). For consistency with that document, EEM terminology was adopted when referring to spatial aspects of the benthic program. Relevant terms are defined below with abbreviations of terms provided in parentheses:

- Reference refers to an area or station that is not exposed to treated effluent.
- Exposure refers to an area or station that is exposed to treated effluent.

- Sampling area (area) is defined as a relatively large area, such as a basin or major arm, of a lake that is exposed to a similar concentration of treated effluent. Examples of sampling areas include reference and exposure areas; exposure areas are often subdivided further into near-field, mid-field, and far-field areas.
- Replicate station (station) is defined as a specific, fixed location within a sampling area, within which several field sub-samples are collected. A station generally corresponds to an area of about 10 metres (m) x 10 m and serves as the unit of replication in statistical analyses.
- Field sub-sample (sample) is defined as the material collected from a defined area, such as the area enclosed by the sampling device, at a randomly selected point within a replicate station. Several field sub-samples are collected at each station to arrive at a representative sample.

6.2.1.2 Study Area

The study area includes the main basin and the northwest arm of Snap Lake, and a reference lake referred to as Northeast Lake (Figures 6-1 and 6-2). Lake 13 was also sampled in 2012 as a provisional second reference lake (Figure 6-3). Gaps in station numbering in Figures 6-1 and 6-2 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.



LEGEND

- BENTHIC INVERTEBRATE MONITORING STATION: MAIN BASIN
- BENTHIC INVERTEBRATE MONITORING STATION: NORTHWEST ARM
- SNAP LAKE MINE FOOTPRINT
- WATERBODY
- DEPTH CONTOUR (m)

REFERENCES

DIGITAL MAP FROM MACKAY LAKE, NORTHWEST TERRITORIES, PRODUCED BY DEPARTMENT OF ENERGY, MINES AND RESOURCES. MAP 75M, ORIGINAL SCALE 1:250,000, NAD 83 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. ALL DEPTH VALUES ARE SHOWN IN METRES
2. MAP MOVED TO NAD83 COORDINATE SYSTEM

PROJECT

DE BEERS
GROUP OF COMPANIES

TITLE

BENTHIC INVERTEBRATE MONITORING STATIONS IN SNAP LAKE, 2012

Golder Associates

PROJECT	13-1349-0001.4300		FILE No.	13134900014300A6-1	
DESIGN	TD	2013-03-13	SCALE	AS SHOWN	REV. 0
CADD	JEF	2013-03-30			
CHECK	LMB	2013-04-19	FIGURE: 6-1		
REVIEW	PMC	2013-04-19			

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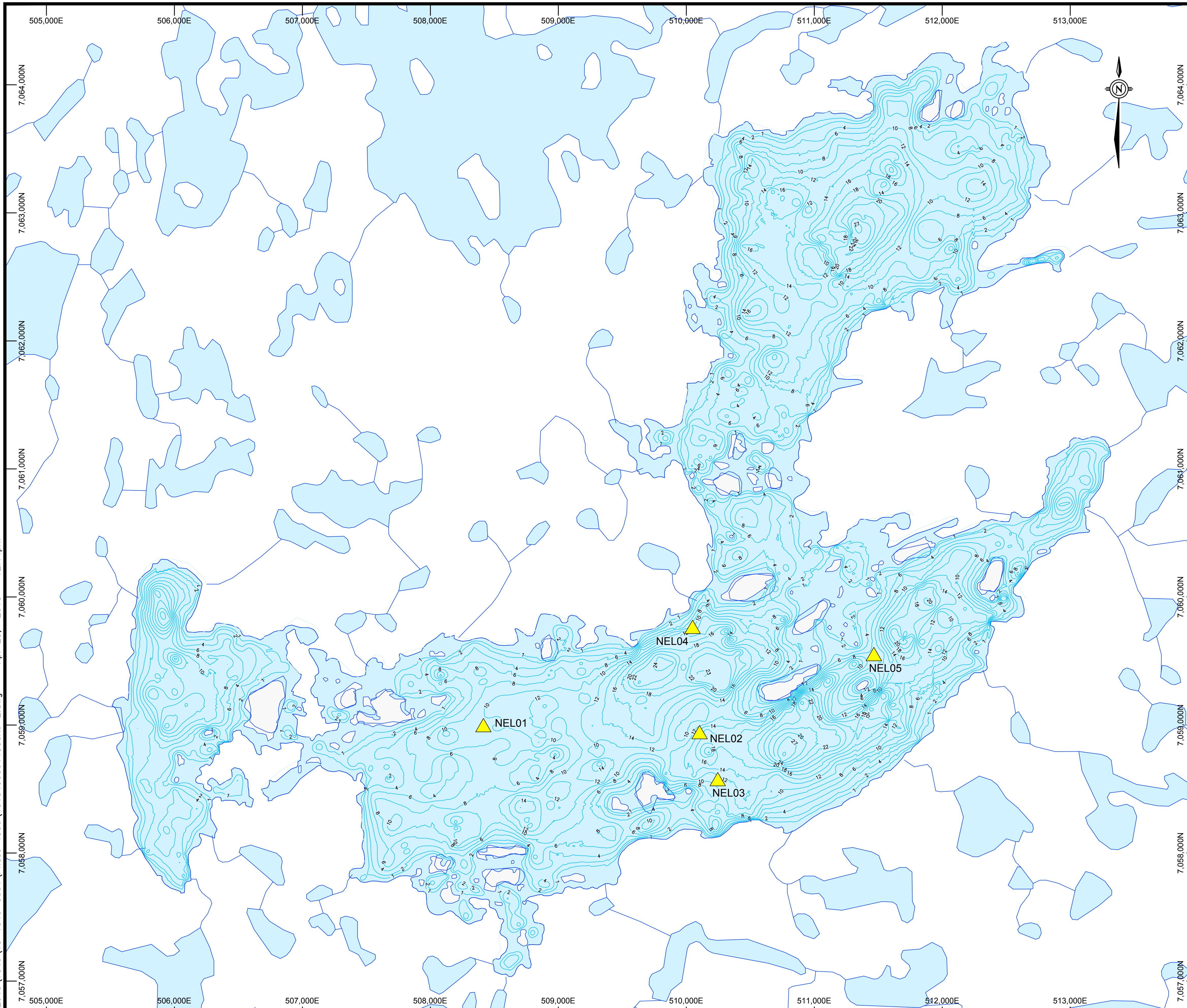


LEGEND

BENTHIC INVERTEBRATE AEMP MONITORING STATION

DEPTH CONTOUR (m)

WATERBODY

REFERENCES

DIGITIZED FROM NTS TOPOGRAPHIC MAP 75 M/10 © 1985 HER MAJESTY THE QUEEN IN RIGHT OF CANADA. DEPARTMENT OF ENERGY, MINES AND RESOURCES. PROJECTION : TRANSVERSE MERCATOR, DATUM : NAD27, COORDINATE SYSTEM : UTM ZONE 12.

REFERENCE 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.

500 0 500

SCALE METRES

PROJECT

DE BEERS
GROUP OF COMPANIES

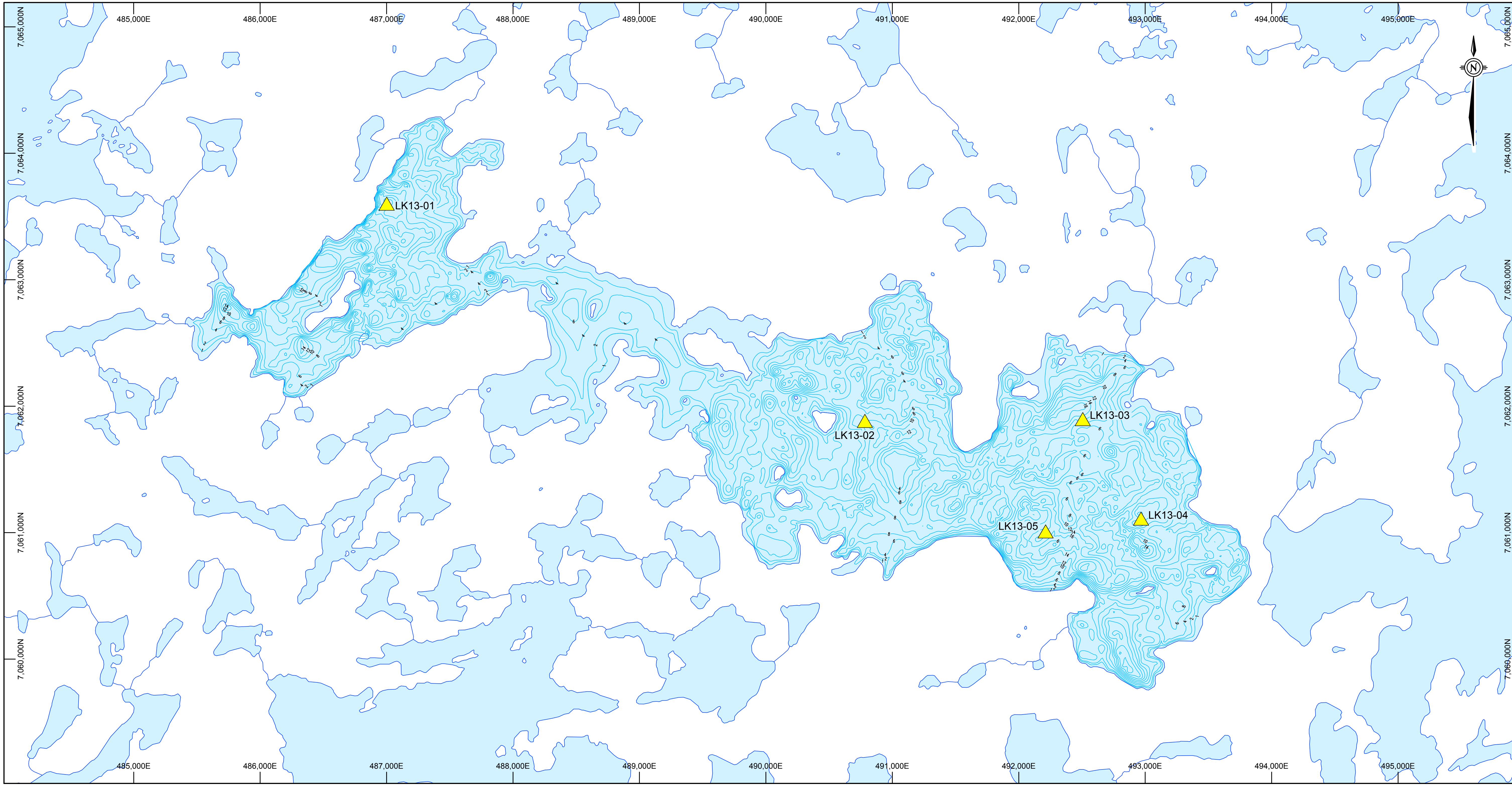
TITLE

BENTHIC INVERTEBRATE MONITORING STATIONS IN NORTHEAST LAKE, 2012

Golder Associates

PROJECT	13-1349-0001.4300	FILE No.	13134900014300A6-2
DESIGN	TD	2013-03-13	SCALE AS SHOWN REV. 0
CADD	JEF	2013-03-30	
CHECK	LMB	2013-04-19	
REVIEW	PMC	2013-04-19	

FIGURE 6-2



500 0 500
SCALE METRES

LEGEND

- BENTHIC INVERTEBRATE MONITORING STATION
- WATERBODY
- DEPTH CONTOUR (m)

REFERENCE

BASE VECTOR DIGITAL CANVEC 1:50K DATA OBTAINED FROM GEOGRATIS ONLINE.

DATUM: NAD83 PROJECTION: UTM ZONE 12

PROJECT

DE BEERS
GROUP OF COMPANIES

TITLE

BENTHIC INVERTEBRATE MONITORING STATIONS IN PROVISIONAL REFERENCE LAKE 13, 2012

Golder Associates

PROJECT	13-1349-0001.4300		FILE No.	13134900014300A6-3
DESIGN	TD	2013-03-13	SCALE AS SHOWN	REV. 0
CADD	JEF	2013-03-30		
CHECK	LMB	2013-04-19		
REVIEW	PMC	2013-04-19		

FIGURE 6-3

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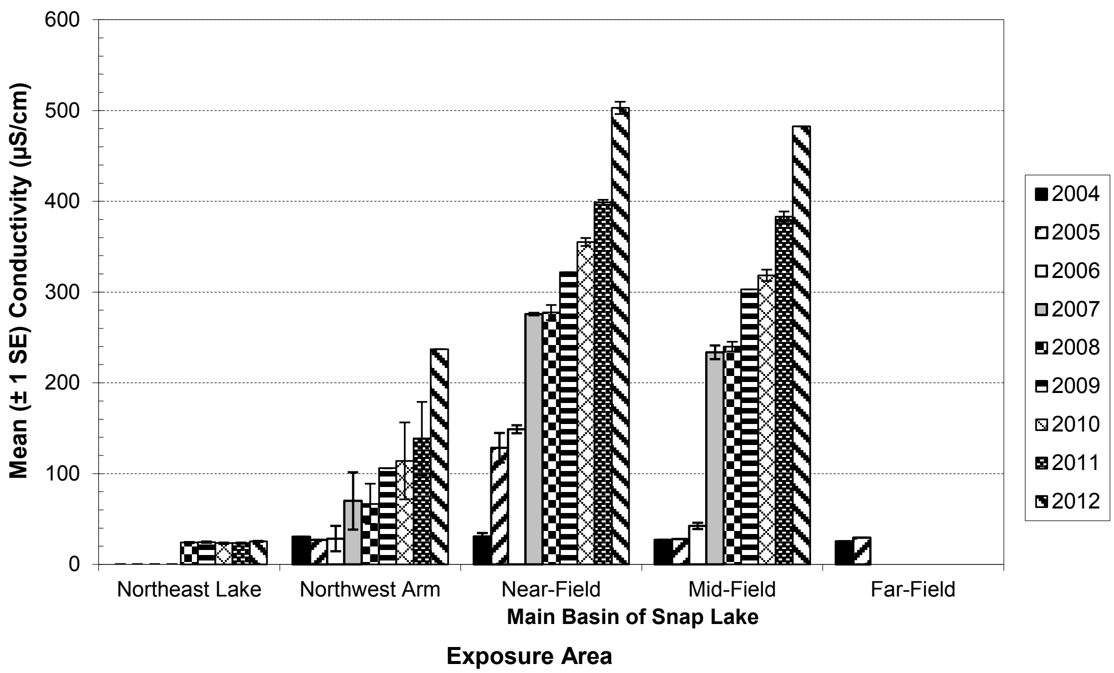
6.2.1.3 Study Design

The original design of the benthic survey was a gradient design, with widely distributed stations throughout Snap Lake, at varying water depths. In 2005, conductivity data indicated that the near-field area was exposed to treated effluent during winter, whereas stations in other lake areas were not exposed (Figures 6-4 and 6-5). As well, water depth at sampling stations was found to influence treated effluent exposure of the lake bottom because of limited vertical mixing under ice. For example, the variation in conductivity apparent in the near-field area in 2005 (Figure 6-4) reflected variation in water depth, with lower exposure at shallower stations, such as SNAP12 and SNAP14.

Figure 6-4 Late-Winter Mean Near-Bottom Conductivity in Each Sampling Area in Snap Lake, 2004 to 2012

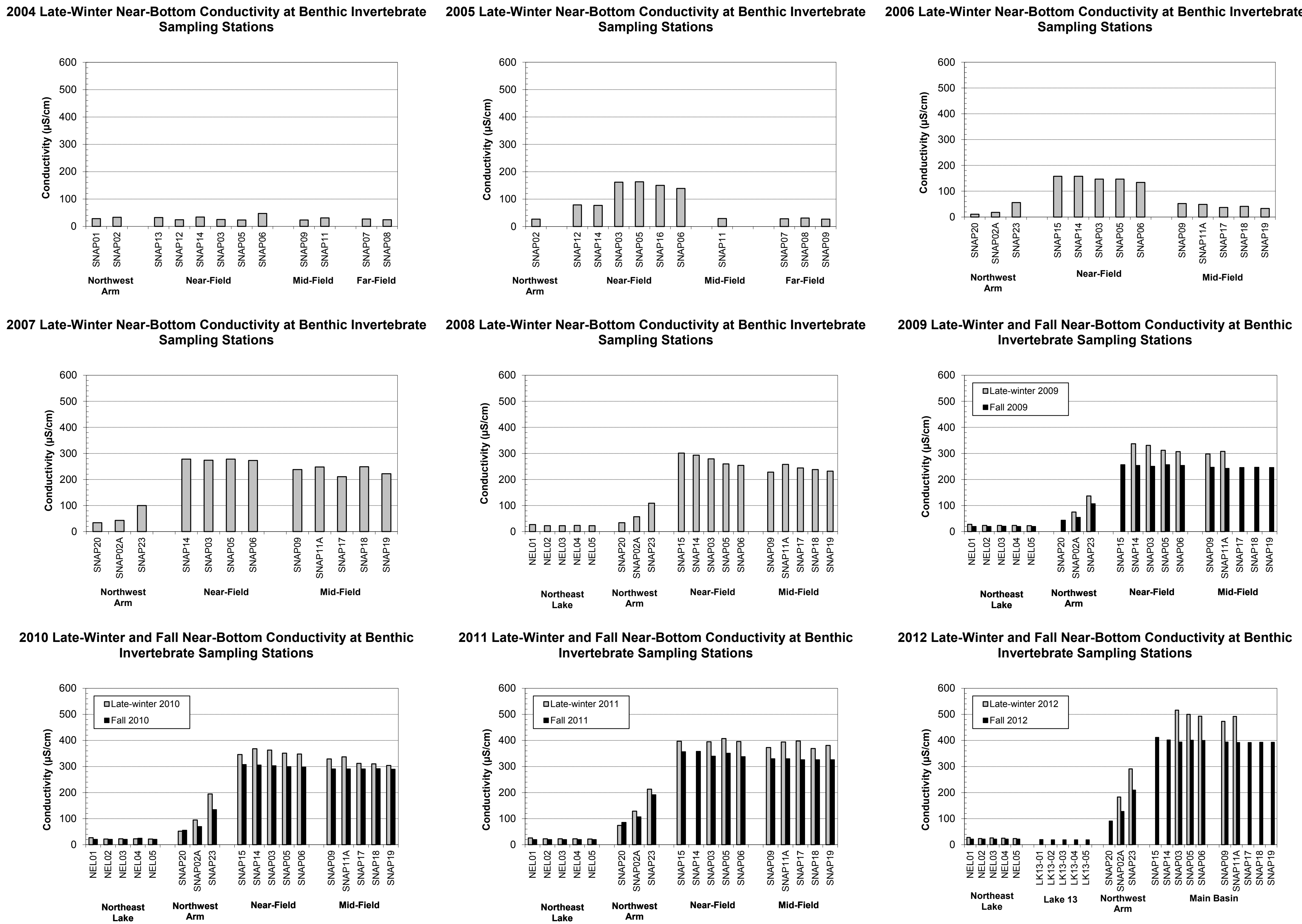


Notes: Area means from 2004, 2005, 2009 and 2012 (northwest arm and mid-field) without standard error bars are calculated based on fewer than three stations per area.

Benthic invertebrate stations were not sampled in the far-field areas from 2006 onwards, due to insufficient water depth at previous sampling locations and the switch to a control/impact design.

µS/cm= microSiemens per centimetre; SE=standard error.

Figure 6-5 Late-Winter and Fall Near-Bottom Conductivity at Benthic Invertebrate Sampling Stations in Northeast Lake, Lake 13 and Snap Lake, 2004 to 2012



Note: Near-field stations are arranged in order of likely treated effluent flow based on lake bathymetry. In 2012, the near-field and mid-field areas were combined into a single area named the main basin due to similar exposure in these two areas.
µS/cm= microSiemens per centimetre; NEL = Northeast Lake; LK13 = Lake 13; SNAP = Snap Lake

Based on the 2005 conductivity data and anticipated future changes in treated effluent concentration throughout the lake, the initial gradient design was changed to a control/impact design for the 2006 survey. As well, sampling depth was standardized to 10 to 15 m, because maximum winter exposure to treated effluent was consistently found in this depth range and natural benthic community variation was low within this relatively narrow depth range. The control/impact design included the northwest arm as a reference area, and near-field and mid-field exposure areas. Sampling in the far-field area was discontinued after 2005 because stations with appropriate water depth were not available in this area. The historical benthic invertebrate sampling locations are shown in Appendix 6A, Figure 6A-1. In 2006, the remaining three sampling areas were exposed to different treated effluent concentrations (Figures 6-4 and 6-5), thereby verifying the appropriateness of the study design in terms of adequately including areas affected by the treated effluent.

Based on the gradual progress of treated effluent exposure across the northwest arm, it has become apparent that this area is of limited use as a reference area. In 2004, which was the baseline year, and 2005, the northwest arm was not exposed to treated effluent and was thus designated as the reference area. However, availability of limited data from this area did not allow statistical comparisons to exposure areas. In subsequent years, only three stations were sampled in the northwest arm, which limited the power of statistical tests comparing reference and exposure areas. In 2006, the first signs of treated effluent exposure emerged in the northwest arm, with the intensity of exposure increasing further from 2007 to 2012 (Figures 6-4 and 6-5). Therefore, although this area remains the least exposed to treated effluent, it no longer functions as a reference area, but rather as an area subject to a low, but variable level of exposure.

The above issues related to the northwest arm are not expected to affect the ability of the 2012 benthic invertebrate program to answer EAR questions. To compensate for the loss of this reference area, five stations were sampled from 2008 onward in Northeast Lake, a suitable reference lake that was also sampled for water and sediment quality. In 2012, Lake 13 was sampled as a provisional second reference lake. In addition, the near-field area was sampled during a reference year and the mid-field area was sampled in 2006, a year with very low exposure to treated effluent in the mid-field area. Both the near-field and mid-field areas were sampled in two or more subsequent years, which allowed among-year comparisons and examination of temporal trends to evaluate potential effects based on under-ice data. Temporal trends are used to determine whether the benthic invertebrate community has changed over time in the main basin of Snap Lake.

Substantial changes were observed in treated effluent exposure in Snap Lake in late-winter 2007, relative to previous years, which had implications on the study design and data analysis approach. Based on near-bottom conductivity measurements, treated effluent exposure increased substantially in the main basin (near-field and mid-field areas) in 2007 and has gradually increased from 2008 to 2012 (Figure 6-5). In addition, although treated effluent exposure in the mid-field area was low in 2006, it increased to a concentration similar to that in the near-field area in 2007 and has continued to increase (Figure 5-4). The near-field and mid-field areas are now generally similar in terms of treated effluent exposure, despite a different exposure history, and thus are combined into a single area (main basin) for statistical analyses and evaluation of trends.

The design of the 2012 survey allows data analysis using control/impact comparisons. Although some background lake-to-lake variation in benthic community characteristics is expected, Northeast Lake and Lake 13 appear to be suitable reference lakes for comparisons with the exposure area of Snap Lake. Northeast Lake and Lake 13 are of similar size and morphology, and water and sediment quality in Northeast Lake are similar to those in Snap Lake under baseline conditions. In addition, the major taxonomic groups present are similar among the lakes with the exception of amphipods (Amphipoda) which are only present in Northeast Lake and biting midges (Ceratopogonidae) which are only present in Lake 13, based on currently available data. Because these taxa are common and widespread, it is possible that they are present in all lakes but have not been observed to date, because a larger sampling effort would be required to consistently detect them. Benthic invertebrate community summary variables for NEL were generally within the historical ranges of variability observed in Snap Lake; however, summary variables were generally higher in Lake 13 compared to both NEL and Snap Lake in 2012.

6.2.1.4 Sampling Methods

Benthic invertebrate samples were collected during the fall open-water program in Snap Lake and Northeast Lake from September 5 to 11, 2012 and in Lake 13 from August 18 to 20, 2012.

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% buffered formalin.

Samples were shipped to Jack Zloty, PhD, located in Summerland, British Columbia, for enumeration and taxonomic identification of invertebrates. For the majority of stations in Snap Lake, the six grabs were combined into a single composite sample. At one station within each sampling area, 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. In Lake 13, the six grabs from all stations were processed as discreet samples.

At each station, sediment samples were collected for analyses of sediment chemistry and particle size. These samples were double-bagged and shipped on ice to ALS Laboratory Group in Edmonton, Alberta, for analyses of sediment chemistry (metals, nutrients, and carbon content) and particle size distribution. Sediment quality results are described in Section 4.

6.2.1.5 Sample Sorting and Taxonomic Identification

Samples were processed according to standard protocols based on recommendations in Environment Canada (2012) and Gibbons et al. (1993). Benthic invertebrate samples were first washed through a 500 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 size. Because 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; Wiederholm 1983; Oliver and Roussel 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 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.

6.2.1.6 Supporting Environmental Variables

During the benthic invertebrate survey, the following supporting environmental information was recorded:

- 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) for each station;
- 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 MDS meter and 600 QS multi-parameter probe were 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 stage of some Dipteran taxa are non-benthic. Abundance data received as number of organisms per sample were converted to density data consisting of number of organisms per square metre (organisms/m²). Unusual abundance data were validated before data summary 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 based on mean densities of invertebrates:

- total invertebrate density (organisms/m²);
- 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.

Six additional variables were included for statistical comparisons for the 2012 sampling program. Densities of dominant invertebrates based on the 2012 data set, defined as those accounting for more than 5% of total invertebrates across all stations, were compared across sampling areas. These invertebrates were *Valvata sincera* (snails, Gastropoda:Valvatidae), fingernail clams

(Pisidiidae), and four Chironomidae genera (*Microtendipes*, *Corynocera, Stictochironomus*, and *Procladius*). Together, these taxa accounted for 77% of total invertebrates in the 2012 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 densities of dominant taxa, except *Procladius* 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 total density, *Microtendipes* density, *Corynocera* density, S*tictochironomus* density, and *Procladius* 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 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 exception was *Stictochironomus* density where transformations did not eliminate the deviations from normality; this variable was tested 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, total organic carbon (TOC), and the percentage of fine sediments, which consists of the silt and clay particle size fractions. Correlations were run using SYSTAT 13.00.05 (SYSTAT Software Inc. 2009) and were considered statistically significant at P <0.05.

Among-Area Comparisons

Among-area comparisons were conducted for fall data. Fall benthic community variables were compared among sampling areas using a one-way analysis of variance (ANOVA) or the Kruskal-Wallis test (Sokal and Rohlf 1995), which is the non-parametric equivalent of ANOVA.

The northwest arm was excluded from statistical comparisons, because of the varying degree of exposure to treated effluent among stations. Station SNAP02A was exposed to treated effluent at low levels and SNAP23 was exposed to treated effluent at moderate levels. Station SNAP20 was exposed to treated effluent in fall 2012 at a low level.

The statistical analyses followed EEM data analysis protocols (Environment Canada 2012). The unit of replication was the station. Variables compared statistically were total density, richness, diversity, evenness, and densities of Pisidiidae, *Valvata sincera*, *Microtendipes, Corynocera*, *Stictochironomus* and *Procladius*. Results of statistical tests were considered significant at P <0.1. The analyses were run using SYSTAT 13.00.05 (SYSTAT Software Inc. 2009). After a significant ANOVA result, the following comparisons were conducted using planned orthogonal contrasts (Sokal and Rolf 1995):

- Northeast Lake and Lake 13 pooled compared to main basin of Snap Lake;
- Northeast Lake compared to Lake 13; and,
- Northeast Lake compared to main basin of Snap Lake.

Results of contrasts were considered significant at $P<0.03$, after a Dunn-Ŝidák correction for multiple comparisons from the original $P<0.1$ (Sokal and Rolf 1995).

Trends Over Time

Snap Lake main basin means for summary variables were calculated and plotted with the normal range from Northeast Lake overlaid to determine whether any of the variables for a given year were outside the normal range, indicating a difference from the reference areas. Trends over time in the main basin of Snap Lake were also evaluated using these plots.

Statistical Power

For benthic invertebrate monitoring, the recommended critical effect size is ±2 standard deviations (±2 SD), estimated from reference area data (Environment Canada 2012). Using this effect size and a significance level of $\alpha=0.1$, 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 verify 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 2012 fall reference 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 set to summarize community structure and evaluate potential differences in community structure between 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 of the two-dimensional configuration was reasonably low (less than 0.2; Clarke 1993). Non-metric multidimensional scaling was run using SYSTAT 13.00.05 (SYSTAT 2009).

Ordination results were presented as a two-dimensional scatter-plot of the sampling stations in ordination space. Since NMDS does not provide an indication of the taxa associated with each dimension, Spearman rank correlation coefficients were generated between scores on each dimension and abundances of the taxa in the biological data set used for the ordination. Spearman rank correlation coefficients were also generated for total density and total richness. Relationships between ordination scores and treated effluent exposure represented by conductivity were evaluated by generating Spearman rank correlation coefficients.

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 Overview of Procedures

6.3.1.1 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 ranging from 98% to 100% in all samples selected for spot-checks (Appendix 6A, Table 6A-5), indicating that the data quality objective was met.

6.3.1.2 Data Entry

In accordance with Golder’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 10 to 15 m (Table 6-1). Water quality parameters varied little with depth, indicating that Snap Lake, Northeast Lake and Lake 13 were well-mixed during the fall benthic invertebrate program.

Fall 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 in fall 2012 (Appendix 6A, Table 6A-4). Conductivity at northwest arm stations SNAP20, SNAP02A, and SNAP23 was also above background at 91 µS/cm, 128 µS/cm, and 210 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 in 2012 was similar to background levels observed in Snap Lake before 2005, at 22 µS/cm at all stations in Northeast Lake and at 19 µS/cm at all stations in Lake 13.

The fall 2012 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 of at least 90% fines, ranging from 94% to 99% fines (Table 6-2). The composition of bottom sediments in Northeast Lake and Lake 13 was similar, ranging from 96% to 99%, and 95% to 99%, respectively. Total organic carbon ranged from 12% to 21% in Snap Lake, from 15% to 18% in Northeast Lake, and from 7% to 9% 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). The TOC values for Lake 13 were lower than in both Snap Lake and Northeast Lake in fall 2012.

Because of the low ranges of variation in water depth and sediment particle size, these variables were not expected to interfere with the analysis of Mine-related effects. However, TOC differences among lakes may be large enough to affect 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	09-Sep-2012	508422	7058996	11	10	12.9	10.1	22	6.6
		NEL02	09-Sep-2012	510092	7058925	11	10	13.2	10.1	22	6.7
		NEL03	09-Sep-2012	510233	7058564	11	10	13.2	9.9	22	6.4
		NEL04	09-Sep-2012	510044	7059745	13	12	13.2	10.0	22	6.7
		NEL05	09-Sep-2012	511466	7059544	11	11	13.1	9.6	22	6.7
	Mean					11	11	13.1	9.9	22	6.6
	Median					11	10	13.2	10.0	22	6.7
	Minimum					11	10	12.9	9.6	22	6.4
	Maximum					13	12	13.2	10.1	22	6.7
Lake 13	-	LK13-01	18-Aug-2012	487001	7063584	12	12	15.2	9.6	20	6.7
		LK13-02	19-Aug-2012	490783	7061866	11	10	15.1	9.9	19	6.8
		LK13-03	19-Aug-2012	492506	7061880	12	12	15.4	9.6	19	6.7
		LK13-04	20-Aug-2012	492967	7061093	10	10	15.3	10.1	19	6.3
		LK13-05	20-Aug-2012	492212	7060992	15	14	15.2	10.1	19	6.8
	Mean					12	12	15.2	9.9	19	6.7
	Median					12	12	15.2	9.9	19	6.7
	Minimum					10	10	15.1	9.6	19	6.3
	Maximum					15	14	15.4	10.1	20	6.8
Snap Lake	Northwest Arm	SNAP02A	05-Sep-2012	503597	7053220	10	9	13.4	10.1	128	7.5
		SNAP20	05-Sep-2012	500823	7052386	15	14	6.8	8.7	91	7.1
		SNAP23	05-Sep-2012	505389	7053353	14	12	13.4	10.0	210	7.4
	Main Basin	SNAP03	05-Sep-2012	507869	7053461	13	13	13.7	9.2	394	7.4
		SNAP05	07-Sep-2012	508417	7052944	13	12	13.3	10.2	401	7.7
		SNAP06	08-Sep-2012	509430	7052587	13	12	12.9	10.6	400	7.7
		SNAP14	07-Sep-2012	507565	7053026	13	12	13.4	10.2	402	7.6
		SNAP15	07-Sep-2012	507384	7052711	10	9	13.4	10.2	412	7.6
		SNAP09	11-Sep-2012	509864	7051674	15	14	12.6	11.0	394	7.4
		SNAP11A	08-Sep-2012	508609	7051753	14	14	13.3	10.3	392	7.6
		SNAP17	08-Sep-2012	508602	7051333	10	10	13.3	10.3	392	7.6
		SNAP18	11-Sep-2012	509177	7051411	12	11	12.5	11.0	393	7.4
		SNAP19	11-Sep-2012	510118	7051802	12	11	12.6	11.0	393	7.5

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						
Snap Lake (continued)	Mean					13	12	12.7	10.2	339	7.5
	Median					13	12	13.3	10.2	393	7.5
	Minimum					10	9	6.8	8.7	91	7.1
	Maximum					15	14	13.7	11.0	412	7.7

Notes: Field water quality data are from near the sediment water interface at the depth indicated in the profile depth column.

- = not applicable; 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 2012

Lake	Area	Station	Maximum Depth (m)	Total Organic Carbon (%)	Sediment Particle Size Gravel (%)	Sand (%)	Silt (%)	Clay (%)	Fines (Silt + Clay) (%)
Northeast Lake	-	NEL01	11	18	0	3	88	9	97
		NEL02	11	16	0	1	87	12	99
		NEL03	11	17	0	4	86	10	96
		NEL04	13	17	0	1	87	12	99
		NEL05	11	15	0	3	83	14	97
	Mean		11	16	0	3	86	11	97
	Median		11	17	0	3	87	12	97
	Minimum		11	15	0	1	83	9	96
	Maximum		13	18	0	4	88	14	99
Lake 13	-	LK13-01	12	9	1	3	79	18	97
		LK13-02	11	8	0	3	79	18	97
		LK13-03	12	7	0	5	83	12	95
		LK13-04	10	8	0	2	79	19	98
		LK13-05	15	7	0	3	78	19	97
	Mean		12	8	0	3	79	17	97
	Median		12	8	0	3	79	18	97
	Minimum		10	7	0	2	78	12	95
	Maximum		15	9	1	5	83	19	98
Snap Lake	Northwest Arm	SNAP02A	10	20	0	2	89	10	98
		SNAP20	15	12	0	2	83	14	98
		SNAP23	14	21	0	2	84	14	98
	Main Basin	SNAP03	13	19	0	1	86	13	99
		SNAP05	13	19	0	2	89	9	98
		SNAP06	13	18	0	3	85	12	97
		SNAP14	13	18	0	1	86	12	99
		SNAP15	10	17	0	2	85	13	98
		SNAP09	15	17	0	5	80	16	95
		SNAP11A	14	18	0	1	88	11	99
		SNAP17	10	16	0	6	83	12	94
		SNAP18	12	18	0	2	88	10	98
		SNAP19	12	17	0	2	87	11	98
	Mean		13	18	0	2	86	12	98
	Median		13	18	0	2	86	12	98
	Minimum		10	12	0	1	80	9	94
	Maximum		15	21	0	6	89	16	99

Note: 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

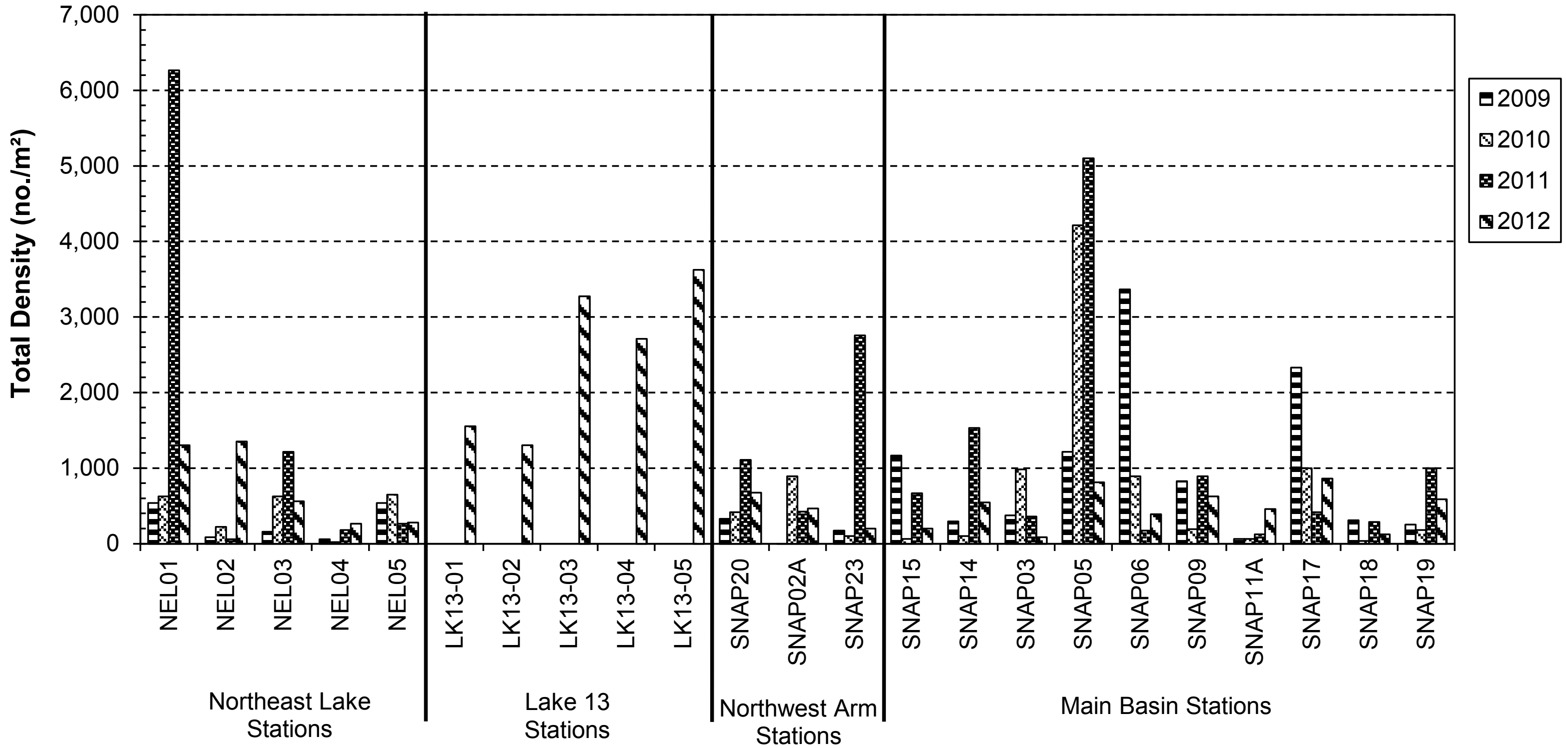
Total invertebrate density was variable but generally low at stations sampled in fall 2012, with a whole-lake mean of 465 organisms/m² in Snap Lake, 753 organisms/m² in Northeast Lake, and 2,494 organisms/m² in Lake 13 (Table 6-3, Figure 6-6; raw data are provided in Appendix 6A, Table 6A-1). Total density ranged from 86 to 863 organisms/m² in Snap Lake, from 266 to 1,353 organisms/m² in Northeast Lake, and from 1,302 to 3,626 organisms/m² in Lake 13. Maximum densities were observed at SNAP17 in the main basin of Snap Lake, at NEL02 in Northeast Lake and at LK13-05 in Lake 13. Density was highly variable among stations within all lakes. Densities at all stations in Lake 13 were higher than at stations sampled in both Northeast Lake and Snap Lake.

Table 6-3 Benthic Invertebrate Summary Variables in Northeast Lake, Lake 13 and Snap Lake, Fall 2012

Lake	Area	Station	Total Density (no./m²)	Total Richness (taxa/station)	Simpson's Diversity Index	Evenness
Northeast Lake	-	NEL01	1,302	13	0.65	0.22
		NEL02	1,353	18	0.85	0.37
		NEL03	561	16	0.84	0.38
		NEL04	266	10	0.63	0.27
		NEL05	281	15	0.87	0.52
	Mean		753	14	0.77	0.35
	Median		561	15	0.84	0.37
	Minimum		266	10	0.63	0.22
	Maximum		1,353	18	0.87	0.52
Lake 13	-	LK13-01	1,554	24	0.91	0.46
		LK13-02	1,302	21	0.90	0.50
		LK13-03	3,273	28	0.66	0.10
		LK13-04	2,712	23	0.84	0.27
		LK13-05	3,626	27	0.77	0.16
	Mean		2,494	25	0.82	0.30
	Median		2,712	24	0.84	0.27
	Minimum		1,302	21	0.66	0.10
	Maximum		3,626	28	0.91	0.50
Snap Lake	Northwest Arm	SNAP02A	468	10	0.73	0.38
		SNAP20	676	13	0.79	0.37
		SNAP23	201	4	0.64	0.70
	Main Basin	SNAP03	86	7	0.76	0.61
		SNAP05	813	17	0.87	0.44
		SNAP06	388	7	0.75	0.57
		SNAP14	547	10	0.81	0.52
		SNAP15	201	7	0.72	0.51
		SNAP09	626	8	0.59	0.30
		SNAP11A	460	11	0.45	0.17
		SNAP17	863	10	0.86	0.71
		SNAP18	122	5	0.54	0.43
		SNAP19	590	11	0.86	0.65
	Mean		465	9	0.72	0.49
	Median		468	10	0.75	0.51
	Minimum		86	4	0.45	0.17
	Maximum		863	17	0.87	0.71

- = not available; no./m² = number per square metre; NEL = Northeast Lake; LK13 = Lake 13; SNAP = Snap Lake

Figure 6-6 Total Benthic Invertebrate Density in Northeast Lake, Lake 13 and Snap Lake, Fall 2009 to 2012



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 30% to 92% of the total density at all stations (Table 6-4, Figure 6-7), with all but four stations having Chironomidae representing greater than 50% of the total density. Pisidiidae were also abundant, accounting for up to 58% of the total density. 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 fingernail clams compared to Northeast Lake in 2012, and a similar proportion compared to Lake 13 in 2012.

In the northwest arm, the benthic community at SNAP23, which had the highest exposure to treated effluent, had a higher proportion of fingernail clams and a lack of Tanypodinae compared to the communities observed at the other two stations (SNAP20 and SNAP02A) (Figure 6-7).

Richness

Richness values in Snap Lake, Northeast Lake, and Lake 13 in 2012 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 compared to these two lakes, ranging from 10 to 18 taxa/station in Snap Lake, from 4 to 17 taxa/station in Northeast Lake, and from 21 to 28 taxa/station in Lake 13 (Table 6-3, Figure 6-8). Richness was significantly positively correlated with total density (r = 0.853; P<0.001). Overall, the fall 2012 richness values in both previously sampled lakes were slightly lower than the ranges observed in 2011.

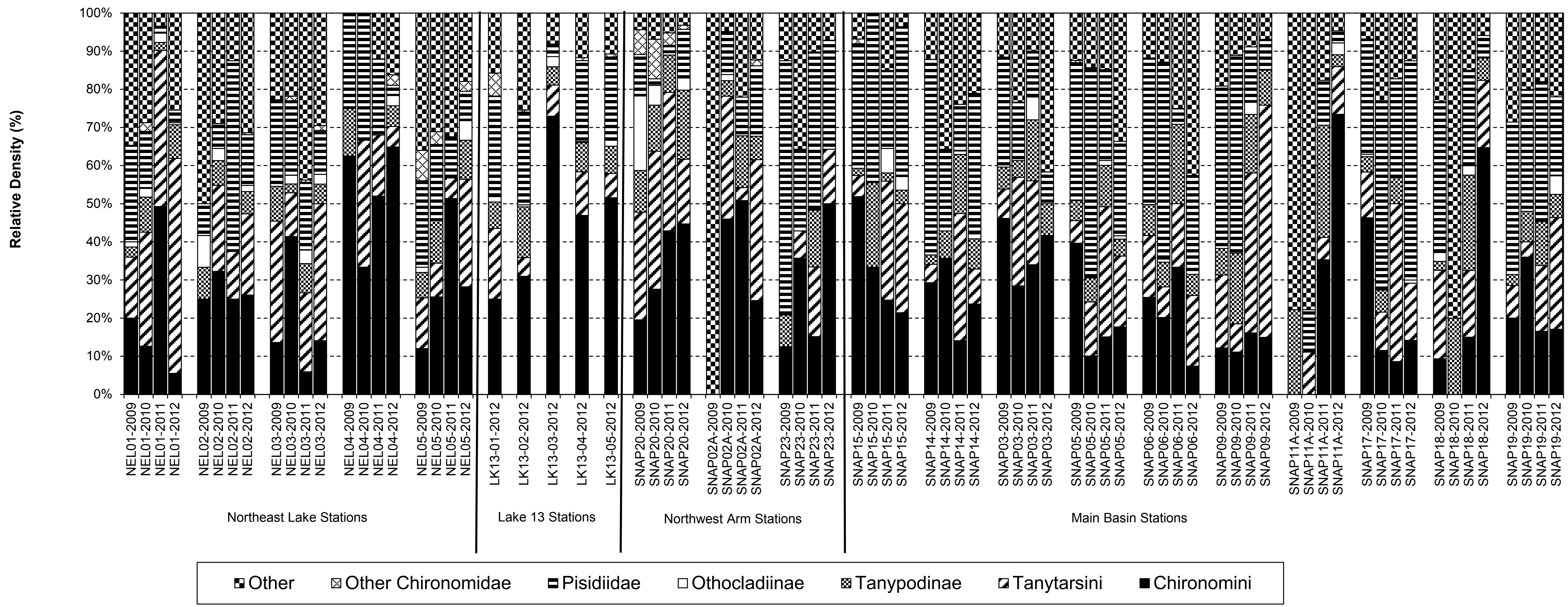
Table 6-4 Relative Densities of Dominant Taxa in Northeast Lake, Lake 13 and Snap Lake, Fall 2012

Taxon	Northeast Lake					Lake 13					Northwest Arm - Snap Lake		
	NEL01 (%)	NEL02 (%)	NEL03 (%)	NEL04 (%)	NEL05 (%)	LK13-01 (%)	LK13-02 (%)	LK13-03 (%)	LK13-04 (%)	LK13-05 (%)	SNAP02A (%)	SNAP20 (%)	SNAP23 (%)
Pisidiidae	3.3	13.3	11.5	2.7	7.7	26.4	24.3	3.1	21.2	21.8	18.5	12.8	28.6
Tanypodinae	8.8	5.9	5.1	5.4	10.3	6.9	13.3	4.8	7.7	7.1	6.2	18.1	0.0
Chironomini	5.5	26.1	14.1	64.9	28.2	25.0	30.9	73.0	46.9	51.6	24.6	44.7	50.0
Tanytarsini	56.4	21.3	35.9	5.4	28.2	18.5	5.0	8.1	11.4	6.3	36.9	17.0	14.3
Orthocladiinae	0.6	1.6	2.6	2.7	5.1	1.4	0.6	2.6	0.3	1.6	0.0	3.2	0.0
Other Chironomidae	0.0	0.5	1.3	2.7	2.6	6.0	0.6	0.2	0.8	0.8	1.5	1.1	0.0
Other	25.4	31.4	29.5	16.2	17.9	15.7	25.4	8.1	11.7	10.7	12.3	3.2	7.1
Total	**100**	**100**	**100**	**100**	**100**	**100**	**100**	**100**	**100**	**100**	**100**	**100**	**100**
Total Chironomidae	71.3	55.3	59.0	81.1	74.4	57.9	50.3	88.8	67.1	67.5	69.2	84.0	64.3

Taxon	Main Basin - Snap Lake									
	SNAP03 (%)	SNAP05 (%)	SNAP06 (%)	SNAP14 (%)	SNAP15 (%)	SNAP09 (%)	SNAP11A (%)	SNAP17 (%)	SNAP18 (%)	SNAP19 (%)
Pisidiidae	8.3	24.8	25.9	36.8	39.3	8.0	3.1	57.5	5.9	20.7
Tanypodinae	8.3	4.4	5.6	7.9	3.6	9.2	3.1	0.0	5.9	6.1
Chironomini	41.7	17.7	7.4	23.7	21.4	14.9	73.4	14.2	64.7	17.1
Tanytarsini	0.0	18.6	18.5	9.2	28.6	60.9	12.5	15.0	17.6	29.3
Orthocladiinae	0.0	0.9	0.0	1.3	3.6	0.0	3.1	0.8	0.0	4.9
Other Chironomidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Other	41.7	33.6	42.6	21.1	3.6	6.9	4.7	12.5	5.9	22.0
Total	**100**	**100**	**100**	**100**	**100**	**100**	**100**	**100**	**100**	**100**
Total Chironomidae	50.0	41.6	31.5	42.1	57.1	85.1	92.2	30.0	88.2	57.3

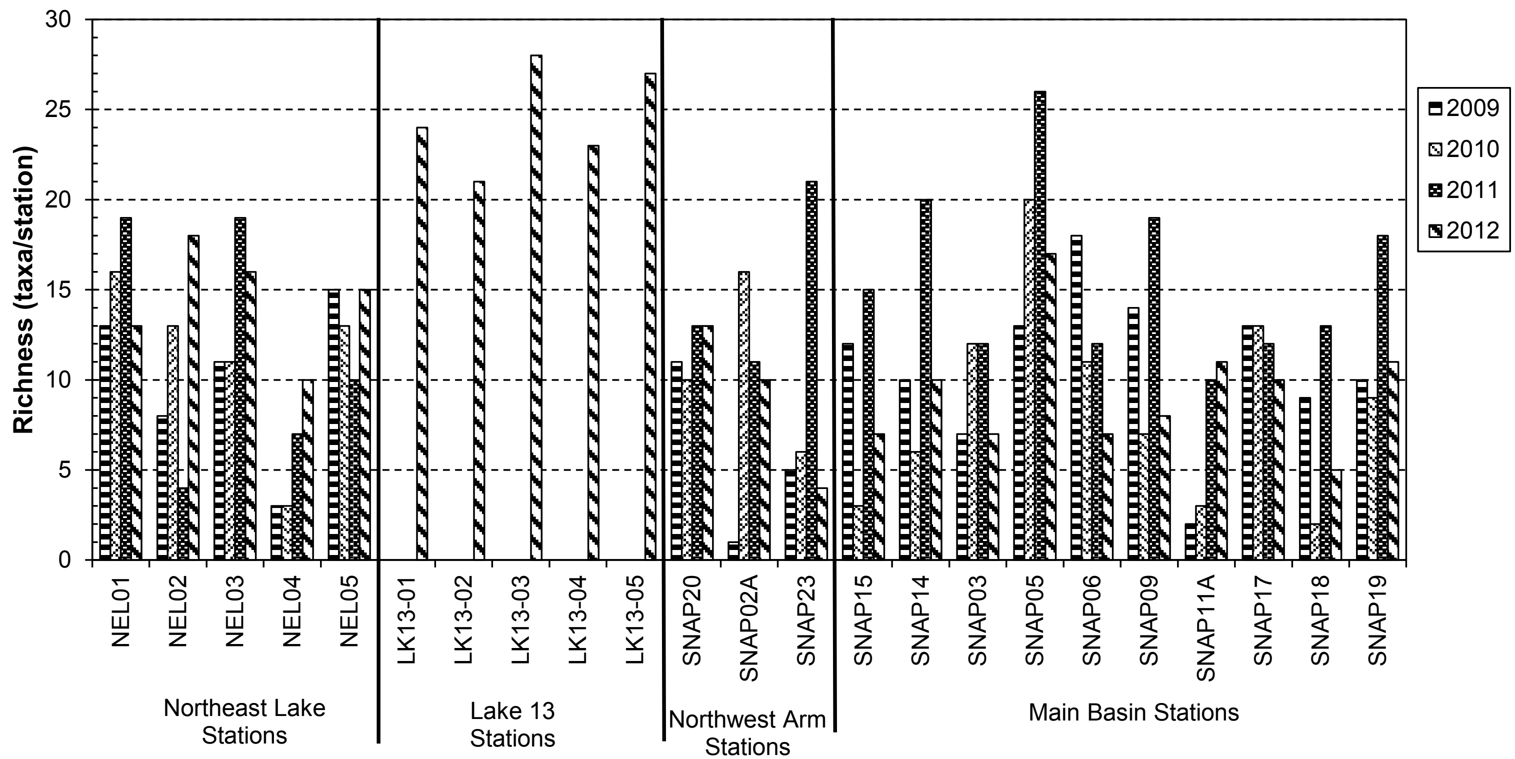
% = percent; NEL = Northeast Lake; LK13 = Lake 13; SNAP = Snap Lake

Figure 6-7 Benthic Invertebrate Community Composition in Northeast Lake, Lake 13 and Snap Lake, Fall 2009 to 2012



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 6-8 Taxonomic Richness of the Benthic Invertebrate Community in Northeast Lake, Lake 13 and Snap Lake, Fall 2009 to 2012



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.

Simpson's Diversity Index

Diversity values varied from 0.45 to 0.87 in Snap Lake in fall 2012, with most values above 0.75 (Table 6-3, Figure 6-9), implying a moderate to high level of diversity. Diversity values in Northeast Lake ranged from 0.63 to 0.87, and in Lake 13 ranged from 0.66 to 0.91 in fall 2012, 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

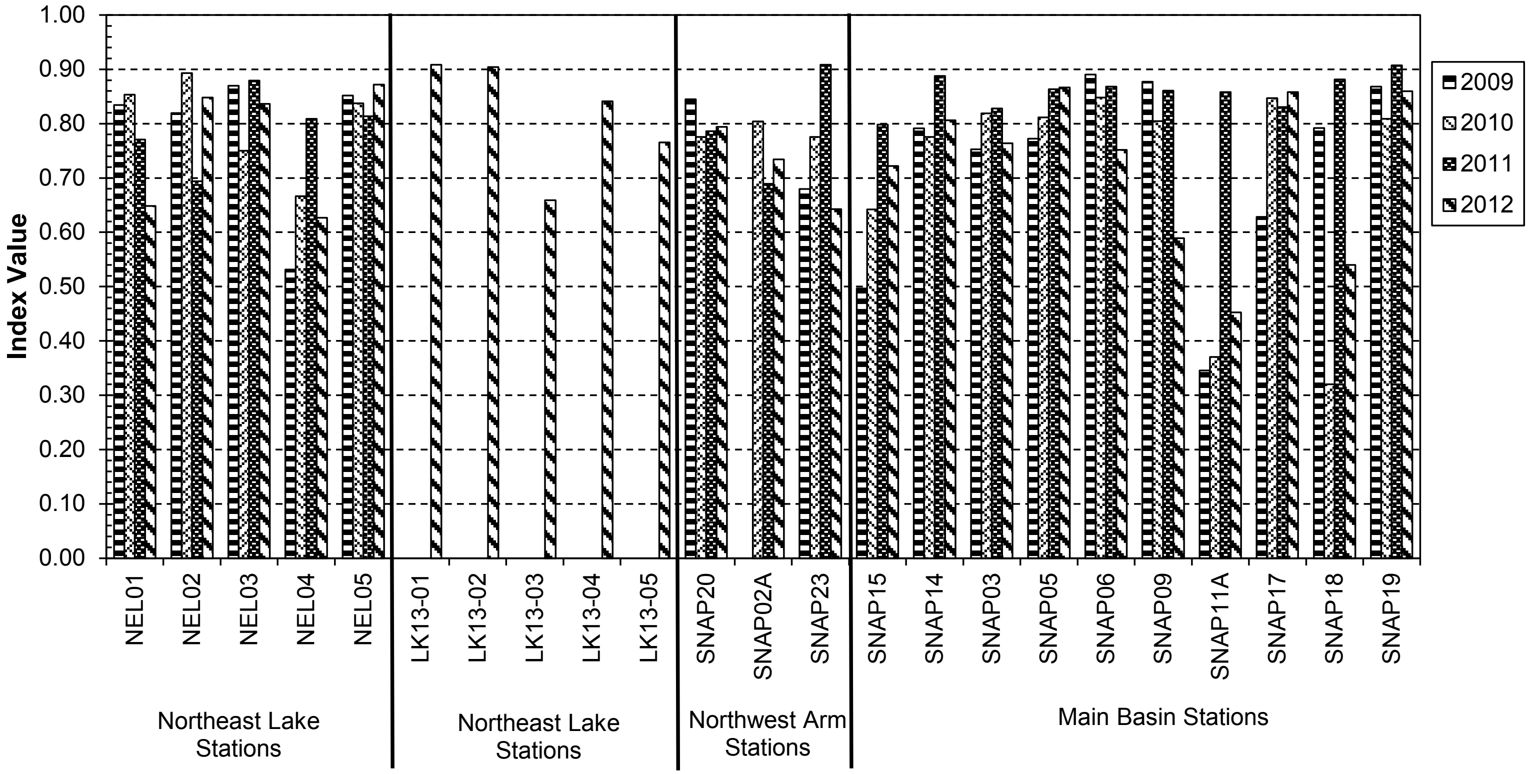
Evenness varied from 0.17 to 0.71, with a mean of 0.49 in Snap Lake in fall 2012 (Table 6-3, Figure 6-10). Snap Lake stations had generally low to moderate evenness. In Snap Lake, high evenness values (greater than 0.60) were generally observed at stations with low total density, and low richness. Evenness was slightly higher in Northeast Lake, ranging from 0.22 to 0.52, with a mean of 0.35, and in Lake 13, ranging from 0.10 to 0.50, with a mean of 0.30 in fall 2012. 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 data were not collected as part of the Lake 13 sampling program. Total benthic invertebrate biomass was highly variable in Snap Lake in fall 2012, ranging from 33.3 to 439.8 milligrams (mg) per station as wet weight (Table 6-5, Figure 6-11; see Appendix 6A, Table 6A-2 for raw data). This represents an approximately 13-fold range in invertebrate biomass among stations. The highest biomass of 439.8 mg was observed at SNAP05 in the main basin of Snap Lake and the second highest biomass of 386.5 mg was observed at SNAP06 also in the main basin of Snap Lake. Biomass in Northeast Lake was also variable, ranging from 53.4 to 910 mg per station 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 2012.

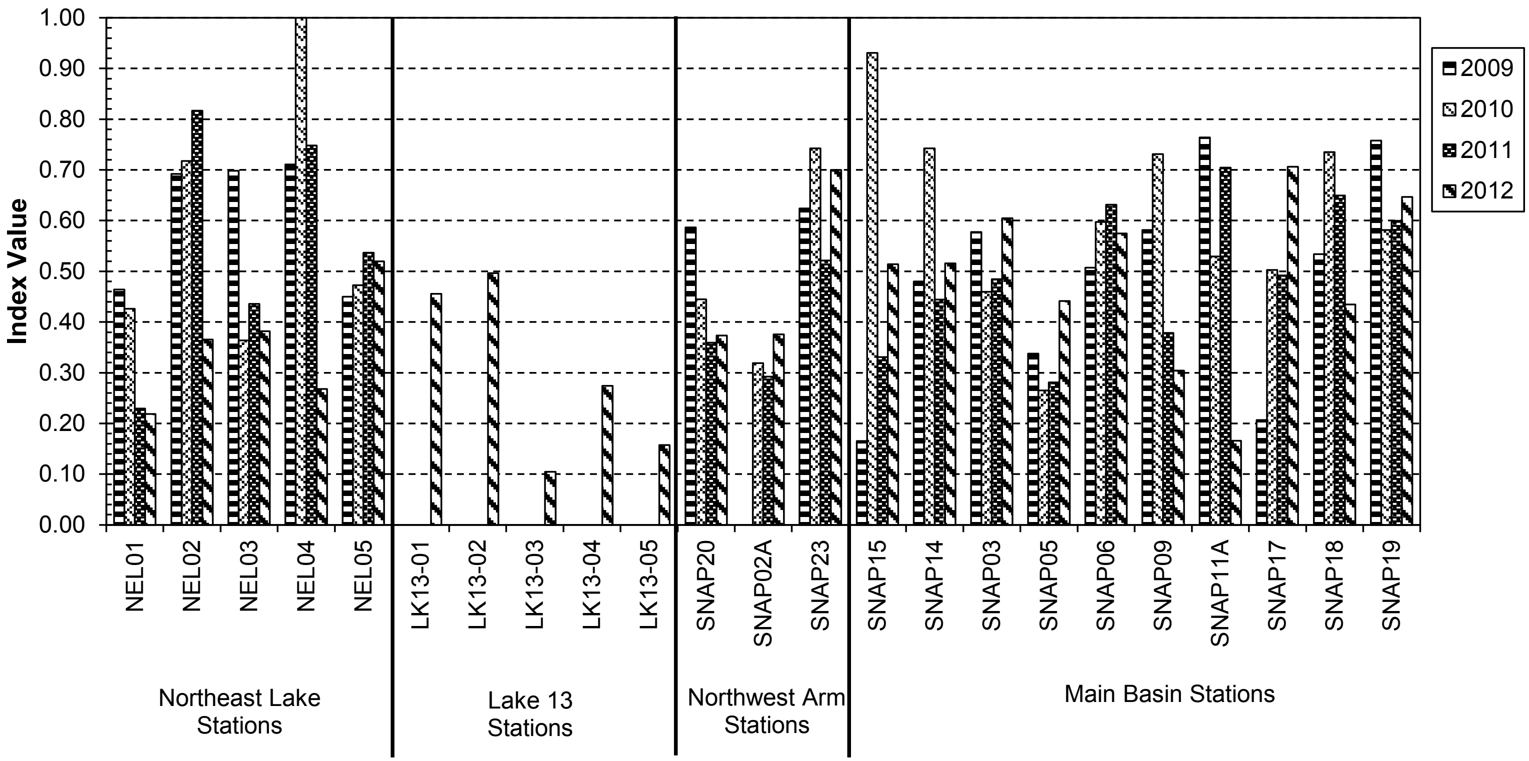
Mean benthic invertebrate biomass was statistically significantly correlated with mean density (r = 0.852; P<0.001). As a result, the spatial pattern in biomass (Figure 6-11) mirrored that of total density (Figure 6-6). The likely reason for this strong correlation is the dominance of Pisidiidae and Chironomidae in the benthic community in Snap Lake, combined with the low degree of size variation within these families.

Figure 6-9 **Simpson's Diversity Index Values for the Benthic Invertebrate Community of Northeast Lake, Lake 13 and Snap Lake, Fall 2009 to 2012**



Note: 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 6-10 Evenness of the Benthic Invertebrate Community of Northeast Lake, Lake 13 and Snap Lake, Fall 2009 to 2012



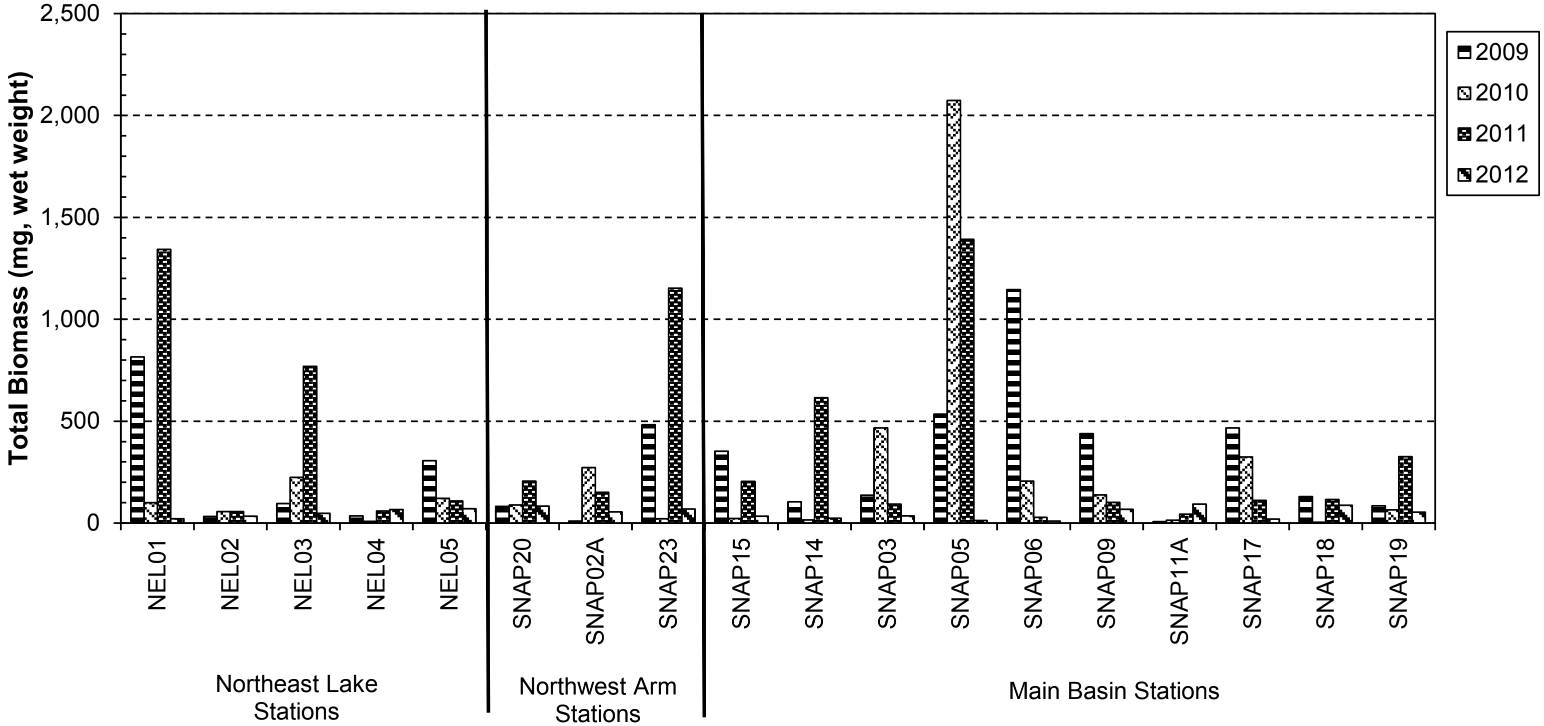
Note: Main basin stations are arranged in order along likely treated effluent flow path based on lake bathymetry.

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 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 6-11 Benthic Invertebrate Biomass in Northeast Lake, Lake 13 and Snap Lake, Fall 2009 to 2012



Note: Near-field stations are arranged in order along the likely treated effluent flow path based on lake bathymetry.
mg = milligram; NEL = Northeast Lake; SNAP = Snap Lake.

Table 6-5 Benthic Invertebrate Biomass in Northeast Lake, Lake 13 and Snap Lake, Fall 2012

	Northeast Lake					North West Arm - Snap Lake		
Taxon	NEL01 [mg, ww]	NEL02 [mg, ww]	NEL03 [mg, ww]	NEL04 [mg, ww]	NEL05 [mg, ww]	SNAP02A [mg, ww]	SNAP20 [mg, ww]	SNAP23 [mg, ww]
Gastropoda/Pelecypoda	617.1	112.4	17.1	14.5	3.7	27.3	23.7	16.3
Oligochaeta	59.7	6.9	16.1	10.3	0.6	15.1	2.2	0.0
Amphipoda	35.6	111.9	23.8	1.5	9.5	0.0	0.0	0.0
Trichoptera	0.0	4.4	0.0	0.0	0.0	0.0	0.0	0.0
Chironomidae	196.4	122.3	52.4	51.5	38.0	53.2	126.5	37.8
Other taxa	1.2	0.0	0.5	0.0	1.6	0.3	0.2	0.3
Total	**910.0**	**357.9**	**109.9**	**77.8**	**53.4**	**95.9**	**152.6**	**54.4**

	Main Basin - Snap Lake									
Taxon	SNAP03 [mg, ww]	SNAP05 [mg, ww]	SNAP06 [mg, ww]	SNAP14 [mg, ww]	SNAP15 [mg, ww]	SNAP09 [mg, ww]	SNAP11A [mg, ww]	SNAP17 [mg, ww]	SNAP18 [mg, ww]	SNAP19 [mg, ww]
Gastropoda/Pelecypoda	6.4	342.6	345.0	126.5	34.1	48.7	8.4	195.6	0.7	55.1
Oligochaeta	14.6	38.7	0.0	13.4	0.0	3.5	0.1	40.9	0.0	12.5
Amphipoda	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Trichoptera	0.0	3.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Chironomidae	11.6	54.4	40.6	43.8	17.4	109.0	115.6	57.2	30.5	81.7
Other taxa	0.7	0.4	0.9	1.6	0.0	0.0	0.1	0.8	3.6	0.9
Total	**33.3**	**439.8**	**386.5**	**185.3**	**51.5**	**161.2**	**124.2**	**294.5**	**34.8**	**150.2**

mg = milligram; ww = wet weight;% = percent; NEL = Northeast Lake; LK13 = Lake 13; SNAP = Snap Lake.

6.4.3 Correlations With Habitat Variables

A statistically significant inter-correlation among habitat variables was detected between percent fines (silt + clay) and TOC, which is expected in lake sediment (Table 6-6). Using the entire data set of 23 stations, total density, richness, Pisidiidae density, *Stictochironomus* density, and *Procladius* density were statistically significantly negatively correlated with TOC. Total density was also significantly negatively correlated with percent fines. Relationships between benthic invertebrate community variables and TOC are 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 (Appendix 6A, Figure 6A-2). As a result, TOC was not included as a covariate in the among-area comparisons. The relationship between total density and percent fines was weak, and thus percent fines also not included as a covariate.

Table 6-6 Spearman Rank Correlations Between Benthic Community Variables and Habitat Variables in Northeast Lake, Lake 13 and Snap Lake, Fall 2012

Variable	Water Depth	Total Organic Carbon	Percent Fines (silt + clay)
Correlations Among Habitat Variables			
Water depth	1	-	-
Total organic carbon	0.136	1	-
Percent fines (silt + clay)	0.116	**0.451**	1
Correlations Between Habitat Variables and Benthic Community Variables			
Total density	-0.133	**-0.706**	**-0.429**
Richness	-0.153	**-0.741**	-0.261
Simpson's diversity index	-0.375	-0.384	-0.136
Evenness	-0.179	0.385	0.027
Microtendipes density	-0.046	-0.411	0.063
Pisidiidae density	-0.259	**-0.486**	-0.264
Corynocera density	-0.216	0.075	-0.289
Stictochironomus density	0.066	**-0.725**	-0.326
Valvata sincera density	-0.171	-0.317	-0.190
Procladius density	0.005	**-0.686**	-0.252

Note: Significant correlations (P <0.05) are shown in **bold** (n =23; r_s = 0.415).
- = not applicable.

6.4.4 Comparison of Snap Lake to Reference Lakes

Summary statistics for the main basin of Snap Lake and Northeast Lake from fall 2012 differed for total density, richness, evenness, and densities of common taxa, with the exception of *Stictochironomus* density and *Valvata sincera* density (Table 6-7). Lake 13 summary statistics were higher than those for the main basin of Snap Lake and Northeast Lake, with the exception of evenness and *Stictochironomus* density, which were lower in Lake 13. Total density and densities of individual taxa were highly variable among stations in Snap Lake, Northeast Lake, and Lake 13, and among lakes (Figures 6-12 and 6-13). High variation in the main basin of Snap Lake resulted from the high densities at SNAP05 and SNAP17. High variation in Northeast Lake resulted from high densities at NEL01 and NEL02. 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 2012

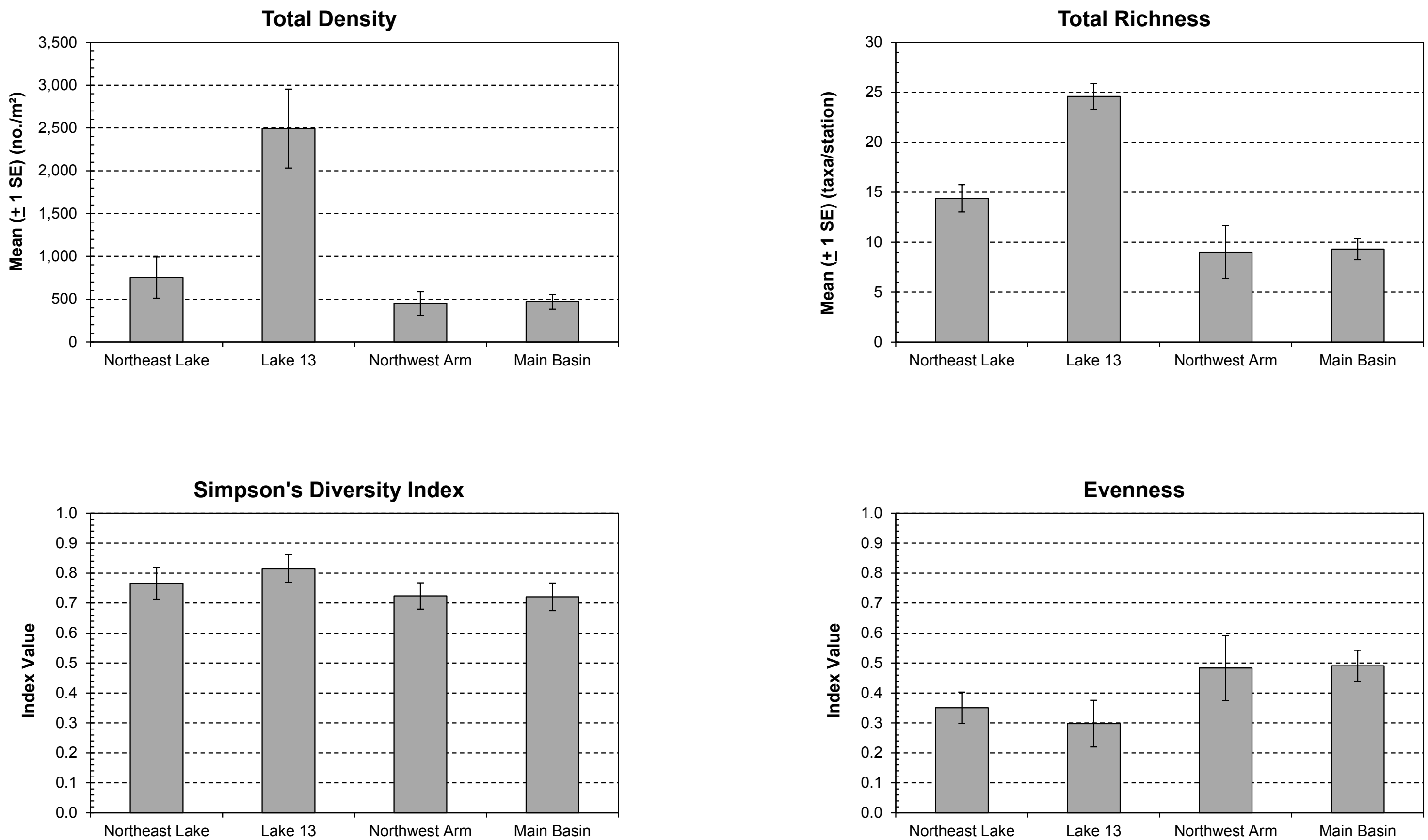
Area	n	Mean	SE	SD	Median	Minimum	Maximum
Total Density (no./m²)							
Northeast Lake	5	753	241	538	561	266	1,353
Lake 13	5	2,494	460	1,030	2,712	1,302	3,626
Snap Lake Northwest Arm	3	448	137	238	468	201	676
Snap Lake Main Basin	10	470	86	272	504	86	863
Total Richness (taxa/station)							
Northeast Lake	5	14	1	3	15	10	18
Lake 13	5	25	1	3	24	21	28
Snap Lake Northwest Arm	3	9	3	5	10	4	13
Snap Lake Main Basin	10	9	1	3	9	5	17
Simpson's Diversity Index							
Northeast Lake	5	0.77	0.05	0.12	0.84	0.63	0.87
Lake 13	5	0.82	0.05	0.11	0.84	0.66	0.91
Snap Lake Northwest Arm	3	0.72	0.04	0.08	0.73	0.64	0.79
Snap Lake Main Basin	10	0.72	0.05	0.15	0.76	0.45	0.87
Evenness							
Northeast Lake	5	0.35	0.05	0.12	0.37	0.22	0.52
Lake 13	5	0.30	0.08	0.17	0.27	0.10	0.50
Snap Lake Northwest Arm	3	0.48	0.11	0.19	0.38	0.37	0.70
Snap Lake Main Basin	10	0.49	0.05	0.16	0.52	0.17	0.71
***Microtendipes* Density (no./m²)**							
Northeast Lake	5	114	57	128	65	7	317
Lake 13	5	914	349	781	892	137	1,856
Snap Lake Northwest Arm	3	72	36	63	101	0	115
Snap Lake Main Basin	10	83	31	97	54	7	338
Pisidiidae Density (no./m²)							
Northeast Lake	5	63	31	69	43	7	180
Lake 13	5	439	117	261	410	101	791
Snap Lake Northwest Arm	3	77	10	17	86	58	86
Snap Lake Main Basin	10	128	47	148	90	7	496
***Corynocera* Density (no./m²)**							
Northeast Lake	5	193	138	309	65	0	734
Lake 13	5	55	39	88	22	0	209
Snap Lake Northwest Arm	3	65	51	88	29	0	165
Snap Lake Main Basin	10	71	35	110	43	0	360

Table 6-7 Descriptive Statistics for Benthic Community Variables in Snap Lake, Northeast Lake, and Lake 13, Fall 2012

Area	n	Mean	SE	SD	Median	Minimum	Maximum
***Stictochironomus* Density (no./m²)**							
Northeast Lake	5	19	10	21	14	0	50
Lake 13	5	224	46	102	230	72	324
Snap Lake Northwest Arm	3	82	82	141	0	0	245
Snap Lake Main Basin	10	24	12	37	0	0	94
***Valvata sincera* Density (no./m²)**							
Northeast Lake	5	42	21	48	14	0	115
Lake 13	5	160	39	86	137	65	295
Snap Lake Northwest Arm	3	5	5	8	0	0	14
Snap Lake Main Basin	10	54	20	64	25	0	180
***Procladius* Density (no./m²)**							
Northeast Lake	5	53	19	42	29	14	115
Lake 13	5	173	25	55	173	108	252
Snap Lake Northwest Arm	3	48	35	60	29	0	115
Snap Lake Main Basin	10	22	6	18	18	0	58

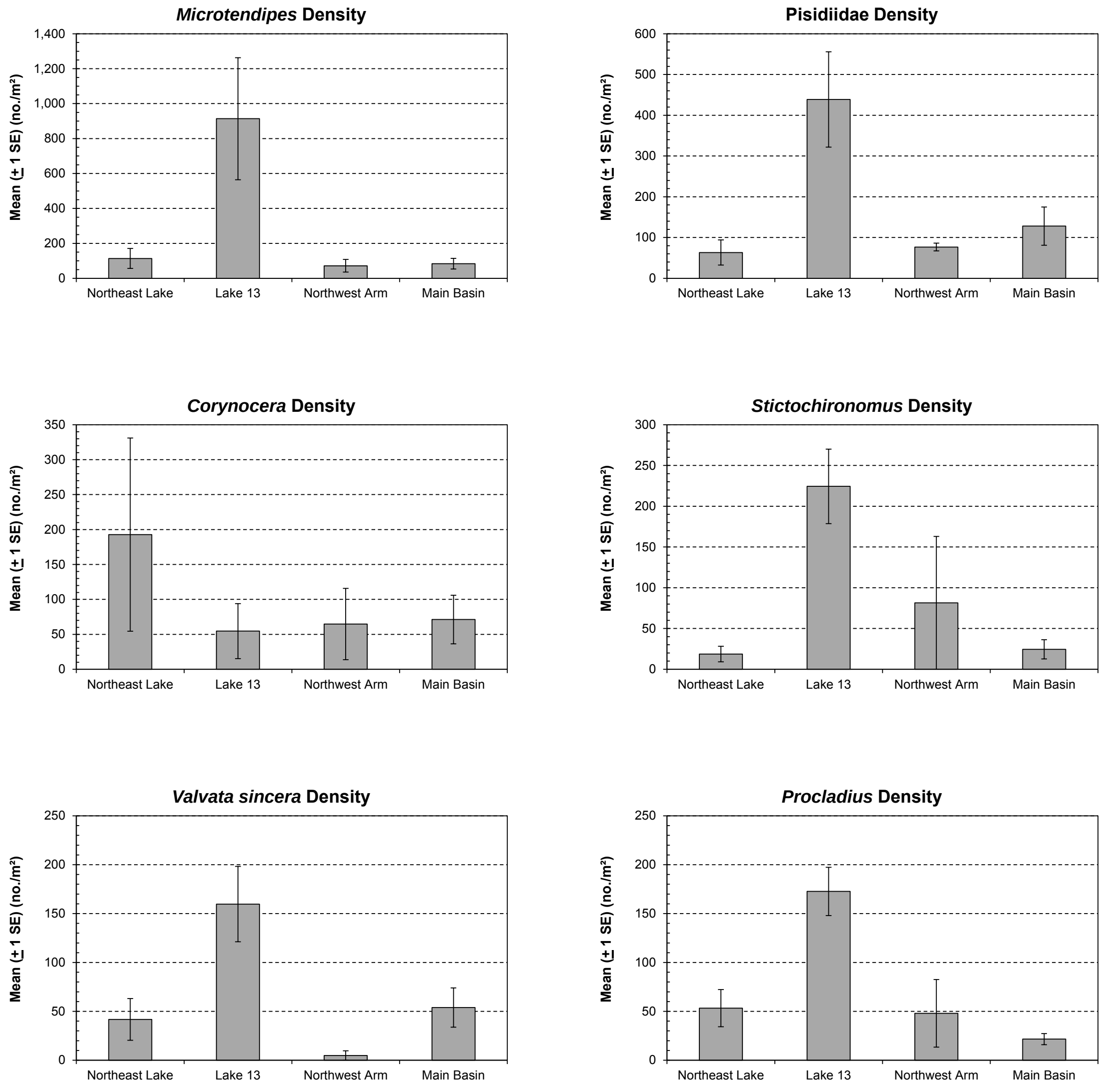
n = number of stations; SE = standard error; SD = standard deviation; no./m^2 = number per square metre.

Figure 6-12 Summary Plots for Benthic Community Summary Variables in Snap Lake, Northeast Lake and Lake 13, Fall 2012



SE= standard error of the mean; no./m² = number per square metre.

Figure 6-13 Summary Plots for Densities of Dominant Benthic Invertebrates in Snap Lake, Northeast Lake and Lake 13, Fall 2012



Note: SE= standard error of the mean; no./m² = number per square metre.

Differences between Snap Lake and Northeast Lake in taxa present were small, with richness ranging from 20 to 26 taxa among areas; however, Lake 13 had a higher richness compared to both lakes with 37 taxa present (Table 6-8). Only minor differences were apparent among lakes in taxa present within the major taxonomic groups in 2012. Oligocheate worms of the sub-family Naidinae were absent from Snap Lake, but present in both Northeast Lake and Lake 13. Amphipoda were only present in Northeast 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.

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, evenness, *Microtendipes* density, Pisidiidae density, *Stictochironomus* density, *Valvata sincera* density and *Procladius* density in 2012 (Table 6-9). Total density, total richness, *Stictochironomus* density and *Procladius* density were statistically significantly lower in the main basin of Snap Lake compared to the pooled reference lakes (Northeast Lake and Lake 13 stations combined). Evenness was significantly higher in the main basin of Snap Lake compared to the pooled reference lakes. However, statistically significant differences were detected between Northeast Lake and Lake 13. Total density, total richness, *Microtendipes* density, *Corynocera* density, *Stictochironomus* density, *Valvata sincera* density and *Procladius* density were statistically significantly lower in Northeast Lake compared to Lake 13.

As a result of the significant differences between the reference lakes, statistical comparisons between Northeast Lake and the main basin of Snap Lake were also conducted. Total richness was statistically significantly lower and *Valvata sincera* density was statistically significantly higher in the main basin of Snap Lake compared to Northeast Lake in 2012 (Table 6-9). The magnitudes of the statistically non-significant differences among lakes were low for both diversity and *Corynocera* density were low (<50%). Magnitudes of differences between Northeast Lake and the main basin of Snap Lake were also low for non-significant results, except for Pisidiidae density, *Corynocera* density and *Procladius* density (>50%) suggesting that the sensitivity of statistical tests comparing these variables among sampling areas was low.

Table 6-8 Presence or Absence of Each Benthic Invertebrate Taxon in Snap Lake, Northeast Lake and Lake 13, Fall 2012

Major Taxon	Family	Subfamily/Tribe	Genus/Species	Northeast Lake	Lake 13	Snap Lake Northwest Arm	Snap Lake Main Basin	Snap Lake Total	All Lakes
Nematoda	-	-	-	X	X	X	X	X	X
Oligochaeta	Lumbriculidae	-	*Lumbriculus*	X	X	X	X	X	X
	Naididae	Naidinae	-	X	X				X
		Tubificinae	-	X	X	X	X	X	X
Hydracarina	-	-	-	X	X		X	X	X
Ostracoda	-	-	-	X	X	X		X	X
Amphipoda	Hyalellidae	-	*Hyalella azteca*	X					X
Gastropoda	-	-	(i/d) [(a)]		X				X
	Valvatidae	-	*Valvata sincera*	X	X	X	X	X	X
Pelecypoda	Pisidiidae	-	(i/d) (a)	X		X	X	X	X
		-	*Pisidium*	X	X	X	X	X	X
		-	*Sphaerium*	X	X		X	X	X
		-	*Pisidium / Sphaerium*		X				X
Trichoptera	Apataniidae	-	*Apatania*		X				X
	Phrygaenidae	-	*Agrypnia*	X	X		X	X	X
Diptera	Ceratopogonidae	-	*Bezzia*		X				X
		-	*Probezzia*		X				X
	Chironomidae	Tanypodinae	*Ablabesmyia*		X	X		X	X
			Procladius	X	X	X	X	X	X
			Thienemannimyia group		X		X	X	X
		Chironomini	(i/d) (a)		X				X
			Chironomus			X	X	X	X
			Cladopelma	X	X				X
			Cryptochironomus	X	X		X	X	X
			Dicrotendipes		X				X
			Microtendipes	X	X	X	X	X	X
			Pagastiella	X	X		X	X	X
			Parachironomus	X	X				X
			Polypedilum		X		X	X	X
			Sergentia			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
			Micropsectra / Tanytarsus		X				X
			Paratanytarsus		X	X	X	X	X
			Tanytarsus	X	X	X	X	X	X
		Orthocladiinae	*Abiskomyia*			X		X	X
			Cricotopus / Orthocladius		X				X
			Heterotanytarsus	X	X				X
			Heterotrissocladius		X				X
			Psectrocladius	X	X		X	X	X
		Diamesinae	*Potthastia longimana* gr.			X		X	X
			Protanypus	X	X	X		X	X
		Prodiamesinae	*Monodiamesa*	X	X				X
Total Taxa				**26**	**37**	**20**	**22**	**28**	**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 2012

Variable	Overall ANOVA Test Result [a] (*P*-value)	Planned Comparisons [b]			Magnitude of Difference			Critical Effect Size		Normal Range (NEL 2009 to 2012)
		NEL and LK13 vs Main Basin (*P*-value)	NEL vs LK13 (*P*-value)	NEL vs Main Basin (*P*-value)	Main Basin from NEL and LK13 (%)	NEL and LK13 (%)	Main Basin and NEL (%)	NEL and LK13 (%)	NEL (%)	
Total density (no./m²)	**<0.0001**	**0.0005**	**0.0006**	0.2855	-71	-107	-38	116	143	0 - 1,314
Total richness (taxa/station)	**<0.0001**	**<0.0001**	**0.0001**	**0.0095**	-52	-52	-35	72	42	2 - 21
Simpson's diversity index	0.4292	0.2491	0.5594	0.5367	-9	-6	-6	40	31	0.58 - 0.99
Evenness	**0.0769**	**0.0289**	0.5981	0.1196	51	16	40	66	66	0.13 - 0.95
Microtendipes density (no./m²)	**0.0056**	0.0365	**0.0077**	0.9130	-84	-156	-27	231	225	0 - 234
Pisidiidae density (no./m²)	**0.0058**	0.1621	**0.0029**	0.4297	-49	-150	102	232	217	0 - 201
Corynocera density (no./m²)	0.5272	0.6312	0.3109	0.3303	-42	112	-63	309	321	- [d]
Stictochironomus density (no./m²) [c]	**0.0059**	**<0.1**	**<0.1**	n.s.	-80	-169	31	303	228	- [d]
Valvata sincera density (no./m²)	**0.0250**	0.1522	**0.0170**	**0.0075**	-46	-117	29	236	229	0 - 146
Procladius density (no./m²)	**<0.0001**	**<0.0001**	**0.0007**	0.0533	-81	-106	-59	166	159	0 - 98

Note: *P*-values representing statistically significant differences are **bold**.

(a) Analysis of Variance (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<0.03 after a Dunn-Ŝidák correction of an original P-value of 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.

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.

6.4.5 Statistical Power and Sensitivity

Based on generic power analysis by 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 10 stations, 5 stations and 5 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 0.79 for Simpson's diversity index and 0.96 for *Corynocera* density for five stations per area. This is a conservative estimate because there were ten stations sampled in the main basin of Snap Lake, compared to five stations in both Northeast Lake and Lake 13. Power for Simpson's diversity index (0.79) was lower than the intended level of power 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 in 2004 to 2006 for Snap Lake, in 2008 to 2012 for Northeast Lake, and in 2012 for Lake 13 were summarized for the five 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 2012 data for Northeast Lake, and 2012 data for Lake 13 was largest for total density, ranging from 70% to 332%, and smallest for diversity, ranging from 9% 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 2012 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 richness and evenness. These differences are due to addition of Lake 13 data in 2012. 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 have re-appeared in 2012 and an increase in the critical effect size for evenness occurred based on the addition of Lake 13 data in 2012.

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 generally 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 2012

Variable	Year	n	Mean	Median	Range	SD	2 SD (%)[a]
Total Density (no./m^2)	2004 - Late Winter	8	948	943	362 - 1,377	332	70
	2005 - Late Winter	5	742	800	373 - 1,290	381	103
	2006 - Late Winter	6	762	670	186 - 2,014	668	175
	2007 - Late Winter	2	366	-	237 - 496	-	-
	2008 (NEL) - Late Winter	5	796	323	194 - 2,529	982	247
	2009 (NEL) - Fall	5	276	158	58 - 540	243	176
	2010 (NEL) - Fall	5	429	626	22 - 647	289	135
	2011 (NEL) - Fall	5	1,597	266	58 - 6,266	2,650	332
	2012 (NEL) - Fall	5	753	561	266 - 1,353	538	143
	2012 (LK13) - Fall	5	2,494	2,712	1,302 - 3,626	1,030	83
Richness (no. of taxa)	2004 - Late Winter	8	13	13	9 - 16	3	43
	2005 - Late Winter	5	13	12	10 - 17	3	51
	2006 - Late Winter	6	12	12	10 - 18	3	49
	2007 - Late Winter	2	10	-	8 - 12	-	-
	2008 (NEL) - Late Winter	5	13	13	10 - 16	3	46
	2009 (NEL) - Fall	5	10	11	3 - 15	5	94
	2010 (NEL) - Fall	5	11	13	3 - 16	4	73
	2011 (NEL) - Fall	5	12	10	4 - 19	7	117
	2012 (NEL) - Fall	5	14	15	10 - 18	3	43
	2012 (LK13) - Fall	5	25	24	21 - 28	3	24
Simpson's diversity index	2004 - Late Winter	8	0.82	0.85	0.66 - 0.89	0.07	18
	2005 - Late Winter	5	0.82	0.81	0.73 - 0.90	0.06	15
	2006 - Late Winter	6	0.82	0.86	0.63 - 0.88	0.10	23
	2007 - Late Winter	2	0.72	-	0.57 - 0.87	-	-
	2008 (NEL) - Late Winter	5	0.86	0.86	0.81 - 0.91	0.04	9
	2009 (NEL) - Fall	5	0.78	0.83	0.53 - 0.87	0.14	36
	2010 (NEL) - Fall	5	0.80	0.84	0.67 - 0.89	0.09	23
	2011 (NEL) - Fall	5	0.79	0.81	0.69 - 0.88	0.07	18
	2012 (NEL) - Fall	5	0.77	0.84	0.63 - 0.87	0.12	31
	2012 (LK13) - Fall	5	0.82	0.84	0.66 - 0.91	0.11	27
Evenness	2004 - Late Winter	8	0.50	0.48	0.33 - 0.70	0.12	49
	2005 - Late Winter	5	0.48	0.38	0.31 - 0.83	0.21	88
	2006 - Late Winter	6	0.55	0.56	0.25 - 0.77	0.18	67
	2007 - Late Winter	2	0.46	-	0.29 - 0.63	-	-
	2008 (NEL) - Late Winter	5	0.63	0.63	0.33 - 0.90	0.21	67
	2009 (NEL) - Fall	5	0.60	0.69	0.45 - 0.71	0.13	44
	2010 (NEL) - Fall	5	0.60	0.80	0.32 - 0.85	0.26	87
	2011 (NEL) - Fall	5	0.55	0.54	0.23 - 0.82	0.24	87
	2012 (NEL) - Fall	5	0.35	0.37	0.22 - 0.52	0.12	69
	2012 (LK13) - Fall	5	0.30	0.27	0.10 - 0.50	0.17	113

Note: 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 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.

(a) 2 SD expressed as the percentage of the baseline, reference area or Northeast Lake mean.

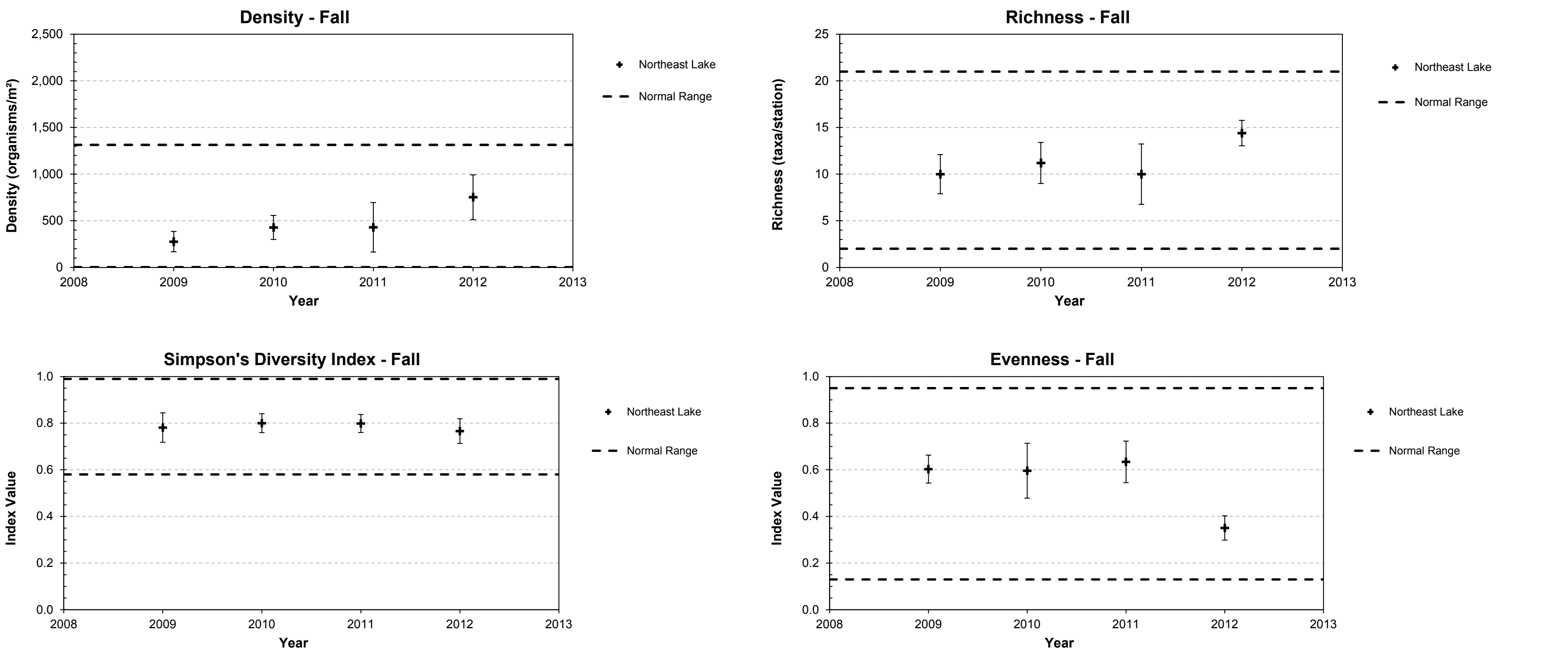
n = number of stations; NEL = Northeast Lake; LK13 = Lake 13; SD = standard deviation;% = percent; µS/cm = microSiemens per centimetre; m = metre; m² = square metre; no. = number; no./m^2 = number per square metre.

6.4.6 Trends Over Time

Total density and total richness have increased from 2009 to 2012 in Northeast Lake (Figure 6-14) compared to a fluctuating trend from 2009 to 2012 and a decrease in 2012 for both variables in the main basin of Snap Lake (Figure 6-15). All values for the main basin of Snap Lake were within the normal range calculated based on Northeast Lake (reference) data from fall 2009 to fall 2012. No trend was present for Simpson's diversity index in either Northeast Lake or the main basin of Snap Lake from 2009 to 2012. No trend in evenness was observed from 2009 to 2011 in Northeast Lake, but a decrease in evenness occurred in 2012. In the main basin of Snap Lake, no trend in evenness was observed from 2009 to 2012. Simpson's diversity index and evenness values for the main basin of Snap Lake were within the normal range.

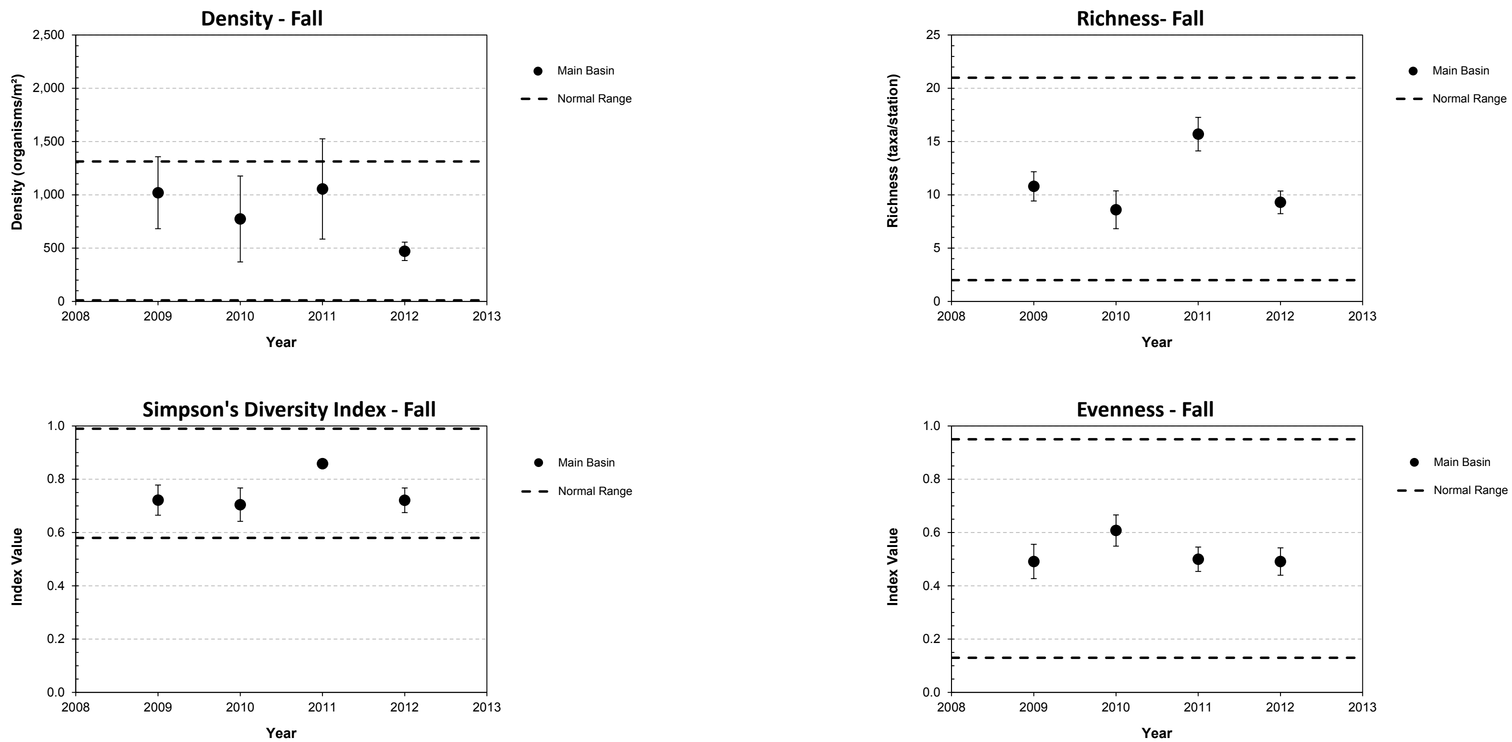
Microtendipes density increased in Northeast Lake (Figure 6-16) from 2009 to 2012 compared to a decrease observed in the main basin of Snap Lake (Figure 6-17). No trend in Pisidiidae density was observed in Northeast Lake compared to a decreasing trend in the main basin of Snap Lake from 2009 to 2012. The decreasing trend in the main basin of Snap Lake has brought the density of Pisidiidae back within the normal range. No trend was present from 2009 to 2012 for *Valvata sincera* density in either Northeast Lake or the main basin of Snap Lake. *Procladius* density has increased in Northeast Lake from 2009 to 2012 compared to no trend in the main basin of Snap Lake over the same time period.

Figure 6-14 Annual Means for Benthic Invertebrate Summary Variables in Northeast Lake from Fall 2009 to Fall 2012



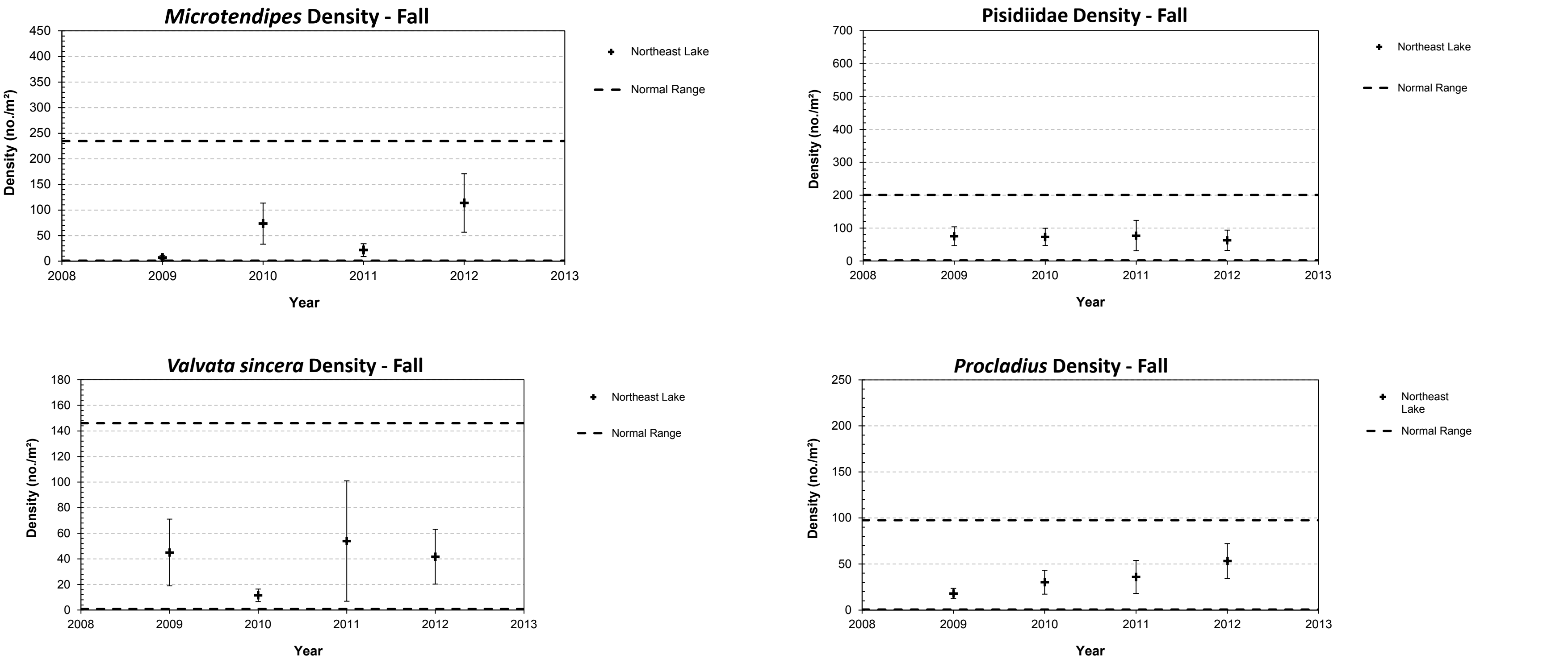
Note: Normal range represents mean ± 2 standard deviations based on pooled reference station means from 2009 to 2012. Error bars represent ± 1 standard error of the mean.

Figure 6-15 Annual Means for Benthic Invertebrate Summary Variables in the Main Basin of Snap Lake from Fall 2009 to Fall 2012



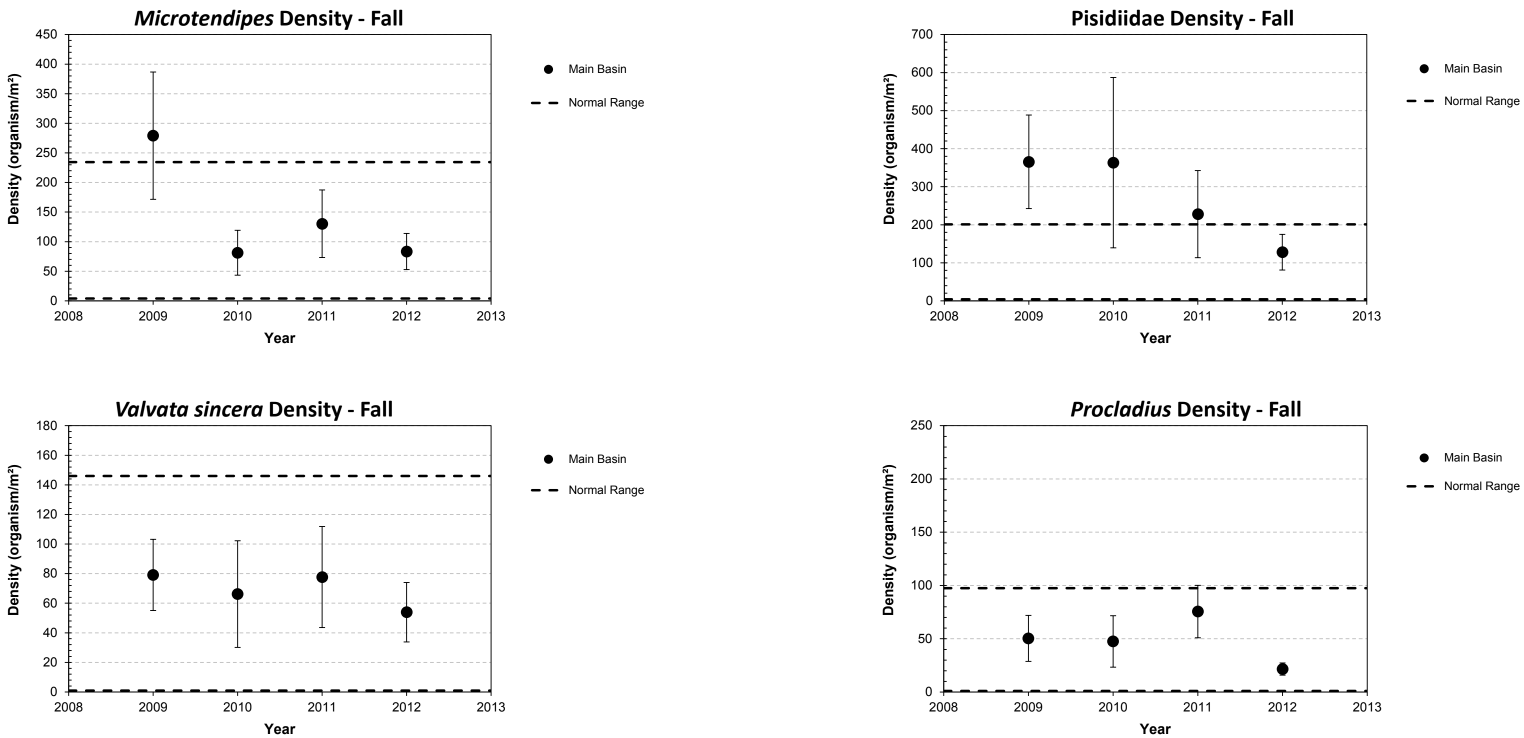
Note: Normal range represents mean ± 2 standard deviations based on pooled reference station means from 2009 to 2012. Error bars represent ± 1 standard error of the mean.

Figure 6-16 Annual Means for Common Taxa in Northeast Lake from Fall 2009 to Fall 2012



Note: Normal range represents mean ± 2 standard deviations based on pooled reference station means from 2009 to 2012. Error bars represent ± 1 standard error of the mean.

Figure 6-17 Annual Means for Common Taxa in the Main Basin of Snap Lake from Fall 2009 to Fall 2012



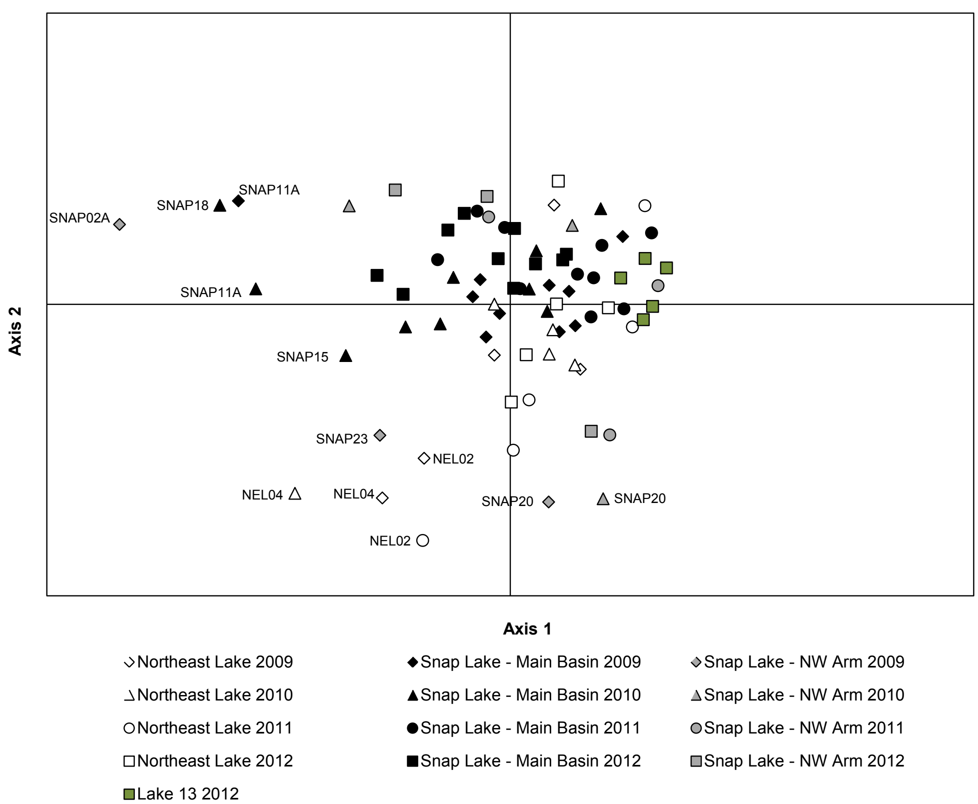
Note: Normal range represents mean ± 2 standard deviations based on pooled reference station means from 2009 to 2012. Error bars represent ± 1 standard error of the mean.

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 0.20, 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 6-18. Each symbol on this figure represents the benthic community of a sampling station; stations with more similar communities are located close to each another.

Total density, total richness, and densities of taxa accounting for 91% of the overall total abundance were highly statistically significantly correlated with Axis 1 scores (*P* <0.001; Table 6-11). Correlations between Axis 1 scores, total density, and total richness are shown in Figure 6-19, and indicate a progression from communities characterized by low abundance and richness on the left side of Figure 6-11 to denser, richer communities on the right side of the figure. The only taxa unique to Axis 2 were *Chironomus, Hyallela azteca*, *Abyskomyia*, *Potthastia*, and Nematoda, which were a minor component of the overall community, accounting for 5% of the total abundance (Table 6-11).

Figure 6-18 Nonmetric Multidimensional Scaling Ordination Plot of the Fall 2009 to Fall 2012 Benthic Invertebrate Data



NW = northwest

Table 6-11 Spearman Rank Correlations Between Nonmetric Multidimensional Scaling Axis Scores, Conductivity, and Biological Variables

Variable	Axis 1	Axis 2
Habitat Variable		
Conductivity	**-0.231***	**0.375*****
Biological Variables		
Total Density	**0.910*****	0.190
Total Richness	**0.902*****	0.111
Procladius	**0.787*****	0.151
Micropsectra	**0.736*****	**-0.321****
Pisidiidae	**0.697*****	0.133
Tubificinae	**0.680*****	0.068
Valvata sincera	**0.657*****	**0.329****
Stictochironomus	**0.642*****	**-0.363****
Tanytarsus	**0.606*****	0.153
Pagastiella	**0.584*****	0.121
Cladopelma	**0.578*****	0.156
Naidinae	**0.490*****	**0.268***
Cryptochironomus	**0.490*****	**0.305****
Psectrocladius	**0.485*****	0.115
Microtendipes	**0.469*****	**0.294****
Corynocera	**0.461*****	**0.518*****
Heterotrissocladius	**0.455*****	-0.092
Ablabesmyia	**0.446*****	0.195
Lumbriculus	**0.439*****	0.003
Monodiamesa	**0.417*****	**-0.328****
Protanypus	**0.412*****	**-0.386*****
Ostracoda	**0.403*****	-0.020
Dicrotendipes	**0.365****	0.111
Paratanytarsus	**0.339****	-0.074
Hydracarina	**0.329****	0.017
Micropsectra/Tanytarsus	**0.314****	0.045
Agrypnia	**0.278***	0.044
Criptopus/Orthocladius	**0.272*****	0.015
Sergentia	**0.270***	-0.108
Cladotanytarsus	**0.267***	0.222
Thienemannimyia	**0.266***	**0.253***
Probezzia	**0.257***	-0.006
Polypedilum	**0.250***	0.137
Parachironomus	**0.249***	0.146
Heterotanytarsus	0.215	-0.015
Chironomus	0.193	**0.297****
Apatina	0.186	-0.036
Bezzia	0.165	-0.067
Zalutschia	0.162	-0.126

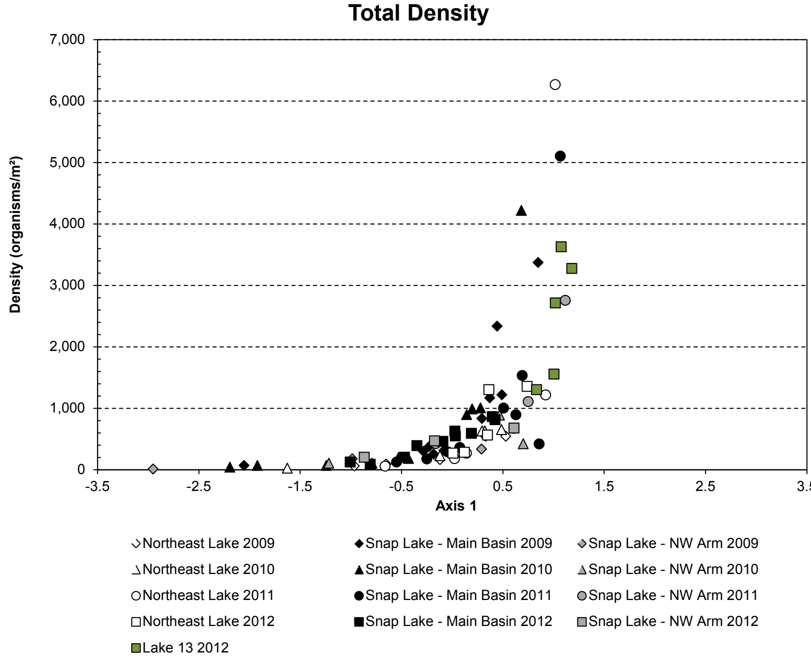
Table 6-11 Spearman Rank Correlations Between Nonmetric Multidimensional Scaling Axis Scores, Conductivity, and Biological Variables

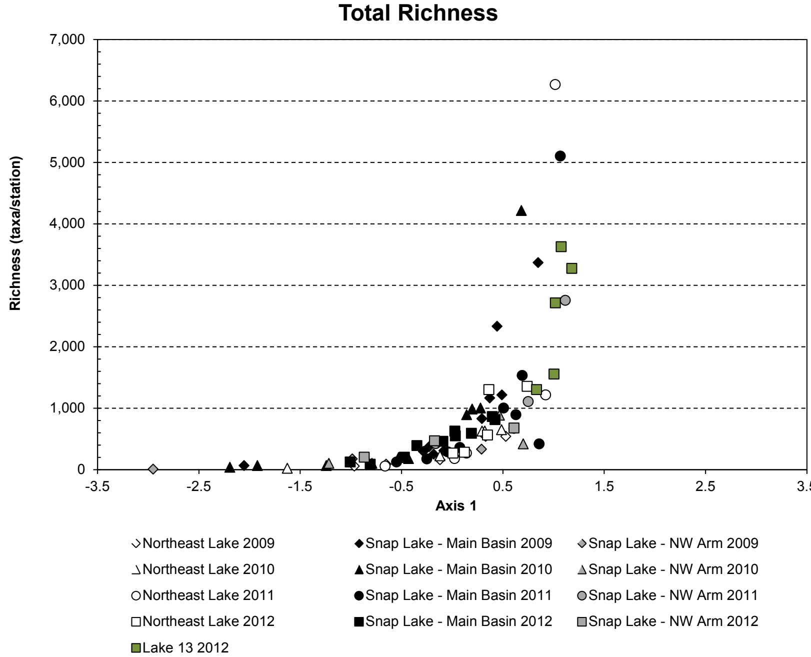
Variable	Axis 1	Axis 2
Hyallela azteca	0.138	**-0.282***
Hydra	0.119	0.155
Gyralus	0.119	0.155
Abyskomyia	0.091	**-0.243***
Potthastia	0.033	**0.242***
Nematoda	-0.003	**0.558*****

n = 77; * = P<0.05; ** = P<0.01; *** = P<0.001.

n=sample size

Figure 6-19 Relationships Between Total Density and Total Richness, and Nonmetric Multidimensional Scaling Axis 1 Scores





m^2 = square metre; NW = northwest.

The Axis 1 vs. Axis 2 ordination plot showed some separation of exposure stations in Snap Lake from reference stations in Northeast Lake and Lake 13 (Figure 6-18), indicating potential evidence of a Mine-related effect. Four stations had lower scores on Axis 1 compared to reference stations. The stations with lower scores on Axis 1 were SNAP11A in 2009 and 2010, and SNAP18 in 2012, 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 as those with low scores on Axis 1 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. 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.

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;
- NEL04 in 2009 and 2010; and,
- NEL02 in 2009 and 2011.

The reasons for the different communities at these stations are unknown.

Consistent with the pattern of separation of exposure stations from reference stations on the ordination plot, statistically significant correlations were found between conductivity, which is an indicator of exposure to the treated effluent, and scores on both Axis 1 and Axis 2 (Table 6-11), indicating a possible Mine-related effect. Conductivity was negatively correlated with Axis 1, which also was highly positively correlated with density and richness.

In summary, NMDS generally ordered stations according to density and richness. 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.

6.5 DISCUSSION

The 2012 Snap Lake benthic invertebrate community program represents the eighth year of monitoring after the discharge of treated effluent began and the sixth year after installation of the permanent diffuser. The 2012 results provide an opportunity to evaluate the effects of the discharge, as well as the appropriateness of the study design.

Understanding of the mixing of treated effluent in Snap Lake is crucial for developing a study design that addresses the objectives of the AEMP (De Beers 2005a). During 2008 to 2012, conductivity measurements in Snap Lake were elevated relative to baseline levels and exhibited little change throughout the entire water column, indicating the effluent was well mixed throughout the water column. Over the past five years of monitoring, exposure to treated effluent has been similar throughout the main basin of Snap Lake. Treated effluent also continued to enter the northwest arm of Snap Lake in 2012. Although its concentration was lower at the northwest arm stations than at main basin stations, treated effluent concentrations have increased in 2012 at the northwest arm stations compared to those observed from 2007 to 2011.

Water quality monitoring during winter 2005 to 2012 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 in TDS, major ions, nutrients, particularly nitrogen, and some trace metals, but sampling area means and whole-lake means of water quality parameters were below benchmarks used in the EAR (De Beers 2002). The concentration of TDS remained below the whole-lake average limit of 350 mg/L specified in the Water Licence (MVLWB 2012). 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). However, there have been increases in concentrations of some water quality parameters relative to baseline levels, especially for TDS, major ions, and nitrogen compounds.

Changes in water quality may influence the benthic community indirectly through altered plankton biomass. Increases in total phytoplankton biomass from 2004 to 2009 followed by declines from 2009 to 2012, and 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 level 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.

The benthic community of Snap Lake in fall 2012 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. Station NEL01 and NEL02 in Northeast Lake, and SNAP05 and SNAP17 in the main basin of Snap Lake had considerably higher total invertebrate density and biomass than all other stations sampled in 2012. Lake 13, which was sampled for the first time in 2012 as a provisional second reference lake, 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, and evenness was higher in Snap Lake compared to Northeast Lake and Lake 13 in 2012.

6.5.1 Statistical Comparisons

Differences between Northeast Lake, Lake 13 and the main basin of Snap Lake during fall 2012 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 total density, total richness, evenness, and common taxa densities for Pisidiidae, *Microtendipes*, *Stictochironomus*, *Valvata*, and *Procladius*. Comparison of the main basin of Snap Lake to the pooled reference lakes (Northeast Lake and Lake 13) found total density, total richness, *Stictochironomus* density, and *Procladius* density were statistically significantly lower in the main basin of Snap Lake compared to the pooled reference lakes. Evenness was statistically significantly higher in the main basin of Snap Lake compared to the pooled reference lakes.

However, statistically significant differences were detected between Northeast Lake and Lake 13. Total density, total richness, *Microtendipes* density, *Corynocera* density, *Stictochironomus* density, *Valvata sincera* density, and *Procladius* density were significantly lower in Northeast Lake compared to Lake 13. As a result of these differences between the reference lakes, statistical comparisons between Northeast Lake and the main basin of Snap Lake were also conducted. Only total richness and *Valvata sincera* density were statistically significantly different between Northeast Lake and the main basin of Snap Lake. Total richness was statistically significantly lower and *Valvata sincera* density was statistically significantly higher in the main basin of Snap Lake compared to Northeast Lake in 2012. This indicates that the majority of differences observed among lakes were between the two reference lakes.

Total organic carbon was also statistically significantly correlated with total density, richness, Pisidiidae density, *Stictochironomus* density, and *Procladius* density in 2012. 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.

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 have declined in 2012. 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.

6.5.2 Trends Over Time

Trends over time also differ in Northeast Lake and the main basin of Snap Lake for some benthic invertebrate summary variables, as follows:

- Total density had an increasing trend from 2009 to 2012 in Northeast Lake compared to a fluctuating trend from 2009 to 2012 and a decrease in 2012 in the main basin of Snap Lake.
- Total richness had an increasing trend from 2009 to 2012 in Northeast Lake compared to a fluctuating trend from 2009 to 2012 and a decrease in 2012 in the main basin of Snap Lake.
- No trend in evenness was observed from 2009 to 2011 in Northeast Lake, but a decrease in evenness occurred in 2012. In the main basin of Snap Lake, no trend in evenness was observed from 2009 to 2012. Evenness values for the main basin of Snap Lake are within the normal range.
- *Microtendipes* density had an increasing trend in Northeast Lake from 2009 to 2012 compared to a decreasing trend in the main basin of Snap Lake.
- 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. The trend has brought the density of Pisidiidae back within the background range for the main basin of Snap 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.

6.5.3 Multivariate Analysis

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 suggest that Mine discharge may have begun to affect benthic community structure.

6.5.4 Provisional Reference Lake 13

Overall, Lake 13 is different than both Snap Lake and Northeast Lake. Total density, richness, *Microtendipes* density, Pisidiidae density, *Stictochironomus* density, *Valvata sincera* density and *Procladius* density were all higher in Lake 13 compared to Northeast Lake. Total organic carbon was also lower in Lake 13 compared to Northeast Lake and Snap Lake. Lake 13 stations also grouped together at the high end of Axis 1 in the NMDS analysis.

6.5.5 Summary

The overall magnitude of the effect on the benthic invertebrate community can be classified as low because statistically significant differences between Northeast Lake and the main basin of Snap Lake were detected for richness, but not total density, or densities of most dominant taxa an in 2012. In 2012, benthic invertebrate summary variables were still within the normal range determined based on data from 2008 to 2012 from Northeast Lake. Taxonomic composition of the community has not changed appreciably compared to baseline conditions. The observed low magnitude is consistent with EAR predictions of a negligible to low effect on the benthic invertebrate community in Snap Lake.

6.6 CONCLUSIONS

6.6.1 In 2012, Was the Benthic Invertebrate Community Affected by the Changes in Water and Sediment Quality in Snap Lake?

Monitoring in fall 2012 detected an effect of low magnitude on the benthic invertebrate community of Snap Lake. 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. The differences in 2012 between Northeast Lake and Snap Lake were smaller than detected during previous years, due to decreases in 2012 in total density, richness and densities of dominant taxa in Snap Lake.

6.6.2 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 2012 was of low magnitude and is consistent with EAR predictions.

6.7 RECOMMENDATIONS

Results of the fall 2012 benthic survey and conductivity data collected in Snap Lake in late winter and fall 2012 were examined to recommend adjustments to the study design for future monitoring under the AEMP. The following recommendation is made for the AEMP benthic invertebrate program:

- The benthic invertebrate community monitoring program should be conducted again in 2013 to determine whether the decreasing trends in total density, *Microtendipes* density, and Pisidiidae density, and the decrease in richness observed in 2012 continue. This recommendation is made because the direction of the effect observed in previous years has reversed in 2012.

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7 FISH HEALTH

7.1 INTRODUCTION

In 2012, De Beers Canada Inc. (De Beers) implemented the field component of the Snap Lake Mine (Mine) Aquatic Effects Monitoring Program (AEMP), as required by the Type A Water Licence MV2011L2-0004. The scope of the AEMP is based on the study design document submitted to the Mackenzie Valley Land and Water Board (MVLWB) in June 2005, which was approved with conditions in July 2005. This section presents the results of the fish health survey conducted under the Mine's AEMP in 2012.

7.1.1 Background

Fish health field programs were conducted at Snap Lake in 1999, 2004, 2005, 2006, and 2009 (De Beers 2002, 2005, 2007, 2010). Baseline data were collected in 1999 (from Snap Lake and an unnamed Reference Lake) and 2004 (from Snap Lake and Northeast Lake) for adult Lake Trout (*Salvelinus namaycush*) and adult Round Whitefish (*Prosopium cylindraceum*) prior to construction of the Mine. The 1999 and 2004 datasets contain adult fish health and tissue metal concentrations and form the baseline dataset for the Mine.

Attempts were made in 2005 and 2006 to non-lethally sample juvenile Lake Trout and Round Whitefish in an effort to reduce overall fish mortality, and to target non-adults, which may be more sensitive to treated effluent from the Mine. The 2005 and 2006 fish survey concluded that it was possible to capture juvenile Lake Trout non-lethally, but not Round Whitefish (De Beers 2006, 2007) and recommended that the fish health study re-focus on finding appropriate study species, considering small-bodied species instead of large-bodied fish species such as Lake Trout and Round Whitefish.

7.1.2 Selection of Fish Species

When choosing a fish species for environmental effects monitoring in aquatic environments, the mobility and residence time of the species relative to treated effluent exposure must be considered. Ideally, a fish species will spend most or all of their life within an area exposed to treated effluent and display limited mobility (i.e., movement over large areas). Species that are migratory, highly mobile, or spend a small proportion of their time in the treated effluent-exposed study area, are not suitable. There are two fish species in Snap Lake that would be suitable as small-bodied fish monitoring species: Slimy Sculpin (*Cottus cognatus*) and Lake Chub (*Couesius plumbeus*).

Slimy Sculpin are a small-bodied fish species with high site fidelity and small home range, making them a preferred study species for environmental effects monitoring studies (Gray et al. 2004). A number of fishing programs have been undertaken in previous years at Snap Lake to collect Slimy Sculpin in sufficient numbers for their consideration as a target species in the AEMP. The preferred method for collection of Slimy Sculpin is backpack electrofishing; however, this fishing method is not feasible in Snap Lake due to the large boulder shoreline habitat present, which is often inaccessible and unsafe for fishing crews. Under-ice fishing efforts using minnow traps were unsuccessful in collecting Slimy Sculpin (Golder 2011a), and subsequent open-water fishing efforts using boat electrofishing, minnow traps, and specialized custom-glass traps were also unsuccessful (Appendix 7A). As a result of these fishing efforts, it was determined that Slimy Sculpin are not a suitable target species for fish health monitoring in Snap Lake.

Lake Chub are a small-bodied fish species with high tolerance to a wide variety of environments, and as such, have the most widespread northern distribution of any North American cyprinid (McPhail and Lindsey 1970). In lakes, Lake Chub tend to occur close to the bottom throughout the littoral zone in the spring, at depths from 0.5 to 10 meters (m), and move closer to shore later in the summer, where they congregate in shallow pools (McPhail and McPhail 2007). In lakes with predatory fish, adult Lake Chub form dense schools and move slowly through the littoral zone during the day. At night, adult Lake Chub move into deeper waters up to 50 m or more from shore and remain close to the surface. Young-of-the-year (YOY) Lake Chub tend to remain solitary and are associated with near-shore shallow waters. Towards the fall YOY Lake Chub join the juvenile population, and remain closely associated with cover during the day. Juvenile Lake Chub are thought to begin to move into exposed littoral areas at night (McPhail and McPhail 2007). Lake Chub spawn in the spring, just after ice-out in northern populations and, while gonadal development is initiated the preceding fall, based on field experience, Golder postulates that gonads continue to develop under-ice during the winter months.

Lake Chub have been captured in large numbers during previous AEMP and fish sampling programs, in Snap Lake and the associated reference lakes (De Beers 2005, 2007). In 2009, the AEMP consisted not only of a lethal large-bodied fish program targeting Lake Trout and Round Whitefish, but also included a small-bodied fish program targeting Lake Chub. Sufficient numbers of Lake Chub were collected in 2009, such that a lethal Lake Chub component was recommended for the 2012 AEMP, and was recommended for inclusion in the 2013 AEMP Design Plan (De Beers 2012). Accordingly, Lake Chub lethal and non-lethal programs were conducted in 2012.

7.1.3 Objectives

The objective of the fish health survey is to determine whether treated effluent has a significant effect on growth, reproduction, survival, and/or condition of fish in Snap Lake. Specific Water Licence conditions applying to the fish health component of the AEMP for the Mine in Water Licence MV2011L2-0004 [Part G, Schedule 6, Item 1a (iii) and 1(d) of MVLWB (2012)] are:

a) *Monitoring for the purpose of measuring Project-related effects on the following:*
iii. fish health;

d) *Procedures to minimize the impacts of the AEMP on fish populations and fish habitat.*

The 2012 fish survey consisted of a lethal and non-lethal small-bodied Lake Chub survey. The fish health component aims to meet the licence conditions by answering the following two key questions:

1. Is fish health affected by changes in water and sediment quality in Snap Lake?
2. Are changes observed in fish health greater than those predicted in the Environmental Assessment Report (EAR)?

The fish health key questions are related to effects predicted in the EAR for the Mine (De Beers 2002) as follows:

- lake-wide increases in total dissolved solids (TDS);
- slight increases in the concentration of hexavalent chromium in the mixing zone and, potentially, in the sediment; and,
- reduced dissolved oxygen concentrations during winter in deeper areas of Snap Lake.

The indirect effects of increased primary production in Snap Lake as a result of treated effluent are also considered by the fish health component. A secondary objective of the 2012 fish health program was to document the health of Lake Chub populations in the waterbodies downstream of Snap Lake, in particular the first lake downstream of Snap Lake (known as Downstream Lake 1).

7.2 METHODS

7.2.1 Field Survey

The fish health survey provides an assessment of whether there are differences in fish health parameters between the exposure and reference lakes. Some of these parameters (e.g., gonad size, liver size) require that the fish be sacrificed through a lethal survey. Other parameters can be measured without harming the fish (e.g., length, total body weight) and are included in both non-lethal and lethal surveys.

7.2.1.1 Study Species and Sample Size

The target sample sizes for the fish health survey were:

- lethal survey: 30 adult male, 30 adult female, and 30 juvenile Lake Chub.
- non-lethal survey: ≥100 (maximum total = 400) Lake Chub from each lake.

7.2.1.2 Sampling Locations

The sampling area for the 2012 fish health study was the main basin of Snap Lake (the exposure lake), Northeast Lake (a reference lake), provisional reference Lake 13 (hereafter referred to as Lake 13), and one lake downstream of Snap Lake, Downstream Lake 1. Fish were collected where suitable Lake Chub habitat was present (Figures 7-1 to 7-4). Attempts were made to fish in the same areas as sampled in the 2004, 2006, and 2009 AEMP programs.

A lethal and non-lethal sampling program was completed for Snap Lake, Northeast Lake, and Lake 13. Due to time constraints, only a non-lethal sampling program was completed for Downstream Lake 1. Snap Lake was accessed by boat from the camp; Northeast Lake, Lake 13, and Downstream Lake 1 were accessed via helicopter whereby boats were slung into each lake for two to six days on each lake.

7.2.1.3 Timing of Sampling

The fish health field program was scheduled to occur in spring, to target pre-spawning Lake Chub, during the period of maximal gonadal development. The 2012 fish health survey consisted of 12 consecutive days of fishing effort, from July 2 to July 14, 2012:

- Snap Lake: July 3 to 4 and July 12 to 14, 2012;
- Northeast Lake: July 6 to 8 and July 13 to 14, 2012;
- Lake 13: July 9 to 11 and July 12 to 13, 2012; and,
- Downstream Lake 1: July 8 to 9, 2012.

7.2.1.4 Fish Collection Methods

Boat electrofishing, minnow traps and hoop nets were used to capture Lake Chub in Snap Lake, Northeast Lake, and Lake 13. Minnow traps and hoop nets were the only fishing methods used in Downstream Lake 1 due to time limitations and the logistic constraints of transporting the electrofisher to the downstream lake.

For each day of fishing on each lake, the following information was recorded:

- time in hours and/or seconds for each fishing effort for each gear type;
- gear specific parameters (e.g., setting for electrofisher);
- water depth of each gear-type set;
- Universal Transverse Mercator (UTM) co-ordinates of each fishing effort;
- substrate type at each fishing location;

- water quality field measurements (dissolved oxygen, water temperature, pH, conductivity, and turbidity), one time daily from each lake;
- numerical documentation of each fishing effort (i.e., a unique effort identification number); and,
- number and species of fish captured and observed.

Boat electrofishing was performed using a Smith Root, Inc. (Vancouver, British Columbia [BC]) Type VI-A electrofisher, in combination with a portable generator and a floating shocking frame attached to an inflatable boat. Electrofisher settings varied over the course of fishing efforts. Direct currents (DC) and alternating currents (AC) were both used; AC was more effective and was used more often than DC. The initial fish captured were inspected for any sign of distress and electrofisher settings were adjusted to avoid stress or injury. Pulse rate remained at approximately 120 pulses per second (pps), voltage ranged from 50 to 1,000 volts (V), voltage output ranged from 50 to 500 V, and current ranged between 0.2 to 8 amps (A). Un-baited minnow traps were set overnight and checked for fish the following day. Similarly, un-baited hoop nets (61 centimetre [cm] diameter hoop, 0.3 cm mesh) were set overnight.

7.2.1.5 Fish Processing

Lethal Survey

External Examinations

Lake Chub captured during the field program were placed in an aerated cooler and transported to an on-site laboratory at the Snap Lake Mine for processing. Incidental mortalities, or fish that did not survive fishing efforts and/or transport to the laboratory, were not included in the lethal fish health survey (but were included in the non-lethal fish survey capture numbers). Any features of the fish that did not appear normal (i.e., wounds, tumours, parasites, fin fraying, gill parasites, or lesions) were reported in detail, and if necessary, submitted for further histopathological analysis. Information on maturity, sex, and overall health were recorded; this information was verified during the internal examination. External examinations of the eyes, gills, thymus, skin, body form, fins, and operculum were conducted on each captured Lake Chub, as per recommendations outlined in Chapter 3 of the *Metal Mining Effluent Regulations Environmental Effects Monitoring Technical Guidance Document* (MMER; Environment Canada 2012). Photographs were taken of any fish with abnormal external features.

Internal Examinations and Organ Collections

Target adult and juvenile Lake Chub were sacrificed by a sharp blow to the head and cervical dislocation (i.e., cutting the spinal cord immediately behind the head) followed immediately by an internal examination. The biological variables collected from lethally sampled Lake Chub were:

- fork length (±1 millimetre [mm]);

- total length (±1 mm);
- total body weight (±0.001 gram [g]);
- physical abnormalities (e.g., tumours, lesions, parasites);
- internal pathology (e.g., liver and kidney colour, fat content);
- parasite weight (if present, ±0.001 g);
- sex;
- stomach contents (% fullness);
- liver weight (±0.001 g);
- whole gonad weight (±0.001 g);
- individual gonad lobe weight (±0.001 g) (females only);
- state of reproductive development (i.e., maturity categories as outlined in Table 7-1);
- carcass weight (±0.001 g); and,
- age (year).

Tissue samples were collected for specialized analyses immediately following or during the internal health assessment:

- gonad histology (each fish);
- fecundity/egg diameter (adult females only);
- liver glycogen, triglyceride, and protein (each fish);
- stomach contents (all fish with more than 50% stomach fullness);
- carcass tissue metals analysis (four male and four female adult fish); and,
- otoliths for aging from each fish (or scales and/or fin rays if otoliths not recovered).

Internal condition was observed and recorded immediately following the opening of the body cavity (i.e., tissue colour and condition). The liver was removed first and as quickly as possible to best preserve the lipids by snap freezing (i.e., storage on dry ice); liver weight was recorded, and the whole liver was placed in a labeled sterile container. During excision of the liver, the gall bladder was observed (if full enough for observation) and its fullness recorded. Stomach fullness was observed and recorded, along with a general description of gut contents and parasite loads. Stomachs with less than or equal to 50% fullness were removed and preserved in a labelled sterile container in 10% buffered formalin for stomach content analyses.

Fish sex and sexual maturity were then recorded as per the maturity stages outlined in Table 7-1, and the gonads were observed for any abnormalities. The whole gonad was removed and weighed, and the individual lobes were weighed separately for female fish. Representative photos of normal and/or abnormal gonads were taken at 10x magnification through a dissecting microscope whenever possible. For males, the total gonad was preserved in 10% buffered formalin in a labelled sterile container for histology. For females, one lobe was processed for histology, while the second gonad lobe was processed for fecundity/egg diameter analysis. Both lobes were preserved in 10% buffered formalin and placed in separate, appropriately labelled sterile containers.

Carcass weight was recorded following removal of the internal organs, but prior to removal of the aging structures. Carcasses consisted of flesh and bone, but not viscera, liver or gonad tissues. Four adult female and four adult male Lake Chub carcasses were frozen and submitted for tissue chemistry analyses. The samples selected included four adult females and four adult males. Sagittal otoliths, scales, or pectoral fin rays were collected for age determination according to the methods outlined by Mackay et al. (1990). Sagittal otoliths are the primary aging structure for Lake Chub; therefore, attempts were made to collect sagittal otoliths from all lethally sampled fish. If two sagittal otoliths could not be collected, a secondary aging structure of scales and/or the left pectoral fin ray were collected.

The variables collected from non-target fish were:

- species;
- physical abnormalities (i.e., wounds, tumours, parasites, fin fraying, gill parasites, or lesions);
- fork length (± 1 mm, if applicable);
- total length (± 1 mm); and,
- total body weight (± 0.001 g).

Table 7-1 Field Maturity Categories Used During the 2012 Snap Lake AEMP Fish Health Survey

Life Stage	Maturity Stage	Definition
1	Unknown (UN)	External examination or unable to determine following internal examination
2	Immature (IM)	Fish has never spawned and will not spawn in the coming season; testes/ovaries transparent, very small and close under the vertebral column, determination of sex difficult
3	Maturing (MA)	Fish has not spawned before, but will spawn in the coming season; gonads developed primarily in the anterior body cavity
4	Seasonal Development (SD)	Sexually mature, has spawned before, gonads developing for coming season
5	Pre-spawning (PR)	Sexually mature, gonads filling ventral cavity, testes white, eggs round and some translucent
6	Ripe (RP)	Roe/milt extruded with very slight pressure on belly
7	Spent (SP)	Spawning completed, reabsorption of residual ovarian tissue not yet completed

Table 7-1 Field Maturity Categories Used During the 2012 Snap Lake AEMP Fish Health Survey

Life Stage	Maturity Stage	Definition
8	Reabsorbing (RB)	Sexually mature but did not spawn; interrupted spawning effort; eggs become atritic (small, hard, white)
9	Resting (RS)	Sexually mature, has spawned; gonads not developing for the coming season; alternate year spawner

Non-Lethal Survey

As described for the lethal survey, all target fish captured were examined externally. Any features of the fish that did not appear normal (i.e., wounds, tumours, parasites, fin fraying, gill parasites, or lesions) were photographed and reported in detail. Information on maturity, sex, and overall health were recorded. External examinations were completed following the recommendations outlined in Environment Canada (2012).

For each fish specimen that was live-released, measurements were:

- species;
- fork length (± 1 mm);
- total length (± 1 mm);
- total body weight (when possible) (± 0.001 g);
- sex (if evident);
- life stage (if evident, otherwise was recorded as unknown); and,
- external health assessment.

This information was recorded on the catch record field data sheet. Measurements were taken in the field and the fish were released near the capture location.

7.2.2 Laboratory

7.2.2.1 Aging

Age determinations were performed by North/South Consultants Inc. in Winnipeg, Manitoba. Lake Chub aging structures submitted were otoliths, fin rays, and scales. Aging structures were examined under a microscope; some fin rays and otoliths were sectioned for ease of observation.

7.2.2.2 Gonad Histology

Gonads were sent for histology analysis to Dr. Mac Law at North Carolina State University, Raleigh, NC, USA. The tissue samples were mounted on slides, sectioned, and stained for microscopic analysis (see Appendix 7B for detailed methods). The histology codes and associated definitions used for categorizing the stages of Lake Chub gonadal development are presented in Table 7-2.

Table 7-2 Laboratory Gonad Histology Codes Used for Lake Chub in the 2012 Snap Lake AEMP Fish Health Survey

Sex	State of Maturity	Histology Code	Definition	Histology
Male	Mature	1A	Sexually mature with normally developing testes	Late spermatogenic: contain primarily spermatozoa (Stage 3 and 4)
	Mature	1B	Sexually mature with normally developing testes, but development retarded compared to 1A	Mid-spermatogenic: contain approximately equal numbers of spermatocytes, spermatids and spermatozoa (Stage 2)
	Immature	2	Immature fish	Pre-spermatogenic: contain only spermatogonia (Stage 6); few to no mitotic cells
	Maturing	3	Male maturing for the first time	Early spermatogenic: contain predominantly spermatocytes and spermatids; groups of mitotic spermatocytes are present (Stage 7 and 1)
	Resting	4	Resting	Contain only spermatogonia (Stage 6); few to no mitotic cells
	Spent	5	Spent testes	Fewer spermatozoa in lumen with prominent ring of spermatogonia lining the tubules (Stage 5)
	Other	6	Testes disorganized	Tubules poorly formed; pockets of asynchronous cells development; residual sperm present
Female	Mature	1A	Sexually mature with normally developing ovary	Advanced maturation: vitellogenic oocytes present (Stages 6 and 7)
	Mature	1B	Sexually mature with oocyte development slightly delayed compared to 1A and some atretic eggs present (>10% but <25%)	Vitellogenesis starts; yolk vesicles present in addition to chromatin nucleolar and perinucleolar oocytes (Stages 3 to 5)
	Immature	2	Immature fish	Chromatin nucleolar oocytes and perinucleolar oocytes only (Stages 1 and 2)
	Maturing	3	Female maturing for the first time (would not have produced viable eggs)	Vitellogenesis starts; yolk vesicles present in addition to chromatin nucleolar and perinucleolar oocytes (Stages 3 to 5)
	Resting	4	Resting	Chromatin nucleolar oocytes and perinucleolar oocytes only (Stages 1 and 2)
	Spent	5	Spawned	Presence of postovulatory follicles and remaining previtellogenic oocytes ± vitellogenic oocytes (Stage 8)
	Other	6	Reabsorbing	Regression: granulation and disintegration of cytoplasm and surrounding follicular layers of oocytes; folded and ruptured remains of oocyes; influx of macrophages and phagocytic follicular cells

% = percent; <= less than; >= greater than.

7.2.2.3 Fecundity and Egg Diameter Estimates

Fecundity analyses and egg diameter measurements were performed by Golder Associates Ltd. (Golder; Saskatoon, SK, Canada). The total number of eggs, or the number of eggs within each lobe, was counted using a dissecting microscope. Average egg diameter was calculated by measuring a subset of 30 eggs. From these data, fecundity was calculated based on one of two equations, as follows:

No subsample:

$$Fecundity = \frac{\#\ of\ eggs\ in\ sample \times total\ ovary\ weight\ (fresh)}{lobe\ weight\ (fresh)}$$ [Equation 7-1]

Subsample:

$$Fecundity = \frac{\#\ of\ eggs\ in\ subsample \times lobe\ weight\ (lab) \times total\ ovary\ weight\ (fresh)}{subsample\ weight\ (lab) \times lobe\ weight\ (fresh)}$$ [Equation 7-2]

where all weights are in grams (g).

Equation 7-1 was used to calculate fecundity when the laboratory did not take a subsample (i.e., analyzed the complete lobe), and Equation 7-2 was used when the laboratory analyzed only a portion, or subsample, of the entire lobe. Egg diameter was also measured in micrometers (μm) and reported as an indicator of egg size.

7.2.2.4 Stomach Contents

Lake Chub stomachs with an estimated fullness ≥50% were sent to Dr. Jack Zloty, Summerland, BC, for enumeration and taxonomic identification of contents. Organisms within the stomach were identified to the genus level using recognized taxonomic keys (Appendix 7D). Organisms that could not be identified to the desired taxonomic level were reported as ‘other’. Individually analyzed stomachs were grouped by lake, sex, and life stage, for an overall summary of presence-absence of organisms to the major taxon level. An estimate of taxon composition within each individual stomach was also determined. From this, the relative percent density of each taxon in stomachs from fish in each lake was calculated.

7.2.3 Data Analysis

7.2.3.1 Approach

Data were analyzed to address the two key questions related to fish health (Table 7-3). Data interpretation considered both the critical effect sizes (CES) consistent with Chapter 1 of the MMER (Environment Canada 2012), and the normal range, calculated as the pooled reference lake mean ± 2 standard deviations. Environment Canada (2012) defines a CES of 25% of the reference area mean for size-at-age, relative gonad size, relative liver size, and age endpoints, and a CES of 10% of the reference area mean for condition. Magnitudes of differences between Snap Lake and the reference lakes that fell below the CES (i.e., less than 25%, or 10% for condition) were considered not biologically significant as they are likely within the range of natural variability for the region and may represent lower environmental risk (Environment Canada 2012).

Table 7-3 Overview of Analysis Approach for Fish Health Key Questions

Key Question	Overview of Analysis Approach
Is fish health affected by changes in water and sediment quality in Snap Lake?	Fish abundance as estimated by CPUE calculated for each gear type for each lake. A lethal and non-lethal small-bodied fish health survey using Lake Chub will measure fish health endpoints related to survival (e.g., age), growth (e.g., size at age), reproduction (e.g., relative gonad size, relative fecundity), and condition (e.g., condition, relative liver size), and will compare these endpoints from Snap Lake with the reference lakes, taking into consideration sex, state of maturity, and parasite presence/absence. Additional analyses from Lake Chub including stomach contents and liver glycogen, triglyceride, and protein concentrations will be undertaken and compared between Snap Lake and the reference lakes.
Are changes observed in fish health greater than those predicted in the EAR?	Fish health endpoints related to survival (e.g., age), growth (e.g., size at age), reproduction (e.g., relative gonad size, relative fecundity), and condition (e.g., condition, relative liver size) measured as part of the small-bodied fish health survey using Lake Chub will be compared to applicable EAR predictions.

CPUE = catch per unit effort; EAR = Environmental Assessment Report .

7.2.3.2 Catch-per-unit-effort

Catch-per-unit-effort (CPUE) provides an estimate of abundance by standardizing catch data according to fishing effort. CPUE was calculated for each species captured, and was summarized by lake and sampling method. The CPUE for electrofishing was calculated as number of fish per 100 seconds effort, and CPUE for minnow traps and hoop nets was calculated as number of fish captured per hour. These calculations provide a measure of relative abundance among sampling lakes by standardizing the catch effort for Snap Lake, Northeast Lake, Lake 13, and Downstream Lake 1.

7.2.3.3 Data Handling

Due to different energetic requirements associated with each sex and state-of-maturity (Environment Canada 2012), fish were grouped by lake, maturity (i.e., adult or juvenile), and sex prior to analyses. Only juvenile and adult Lake Chub undergoing seasonal reproductive development which were free of tapeworms were included in the lethal survey data analyses. Lake Chub were separated based on state-of-maturity, as determined by fish length, for the non-lethal survey.

Data Screening

Data screening was performed prior to performing statistical analyses. Data-checks and maturity confirmations were performed, as follows:

- Fish health data were plotted as box plots (Appendix 7E) and scatterplots to visually examine data for any potential data entry errors or unusual data. Plots included fork length versus total body weight, fork length versus carcass weight, gonad size versus body weight, liver size versus body weight, age versus fork length, age versus total body weight, age versus carcass weight, and age versus gonad size.
- Extreme values, as detected by the visual screening techniques, were removed from the data set only if they were determined to be the result of sampling or measurement errors. All extreme data were checked with field data entry sheets for validity prior to removal from the data set.
- During field surveys, fish more than 50 mm were classified as adults, and fish less than 50 mm were considered to be juveniles (De Beers 2005). This length-maturity categorization was adjusted during the 2012 fish health data screening using the 2012 length-frequency distributions and accompanying data (Appendix 7F). Fish less than 35 mm were considered YOY, fish 35 to 65 mm were considered juvenile, and fish more than 65 mm were considered adults.
- The separation of adults and juveniles was confirmed with further consideration of gonad size and gonad histology results (as per Table 7-2). If there were inconsistencies between gonad histology and field assessment of sex determination, gonad histology conclusions were weighted more heavily.
- Only seasonally developed and ripe fish were included in the adult male and adult female statistical analyses; any immature, maturing, resorbing, resting, or spent fish were not included in the lethal survey analyses.
- Any adult or juvenile fish with tapeworms were excluded from the lethal survey analyses.
- Any adult fish whose sex could not be determined by field assessments and for which gonad histology was lacking was not included in the adult lethal survey analyses.

Data Transformation and Statistical Outliers

Once data were sub-divided based on sex and maturity, but prior to further statistical analyses, tests were performed to determine whether data met parametric assumptions (i.e., were normally distributed and demonstrated equality of variance). All data were $\log_{10}$ transformed and subsequently screened as both raw (untransformed) data and $\log_{10}$-transformed data. Transformations were performed because the majority of biological data do not satisfy the statistical requirement of normality and homogeneity of variance unless log transformed. The goodness-of-fit of each dataset to a normal distribution was tested using a Kolmogorov-Smirnov (K-S) test. The assumption of group variances being equal was tested using a Bartlett's and Levene's test (Systat 2012). A final data screening process was performed for covariate analyses (e.g., gonadosomatic index (GSI), liver somatic index (LSI), condition, and relative fecundity) where linearity was confirmed, and significance of the regression relationships was confirmed prior to statistical testing.

The presence of statistical outliers within a dataset can greatly influence normality and equality of variance, and thus, the type of statistical test (i.e., parametric or non-parametric) that can be performed. Standardized residuals (SR) from linear regression analyses were used as a screening tool for identifying statistical outliers. Unexplained deviations from the regression line were quantified by calculating SR values. The SR considers leverage or influence of an observation on the regression, as well as the magnitude of the residual (Sokal and Rohlf 2012). In brief, if an observation showed high leverage, but had a low residual, this indicates consistency with the regression model and the observation is not unusual within the dataset. Large residuals with low leverage values are not considered unusual, as they do not have notable influence on the regression line. If, however, observations show both high leverage and high residuals (i.e., large SR values), then such observations may affect the slope of the regression line unduly (Sokal and Rohlf 2012). Observations that had SR more than |3| were checked and their validity confirmed. Once confirmed, these observations were considered statistical outliers and were removed from relevant statistical tests. Statistical outliers were clearly identified and the SR value reported for each of the lethal and non-lethal survey analyses. Statistical testing was performed only on outliers-removed datasets following the robust screening procedures and SR outlier identification process described below. All statistical analyses were conducted using the software SYSTAT 13.00.05 (SYSTAT 2009).

7.2.3.4 Descriptive Statistics

Descriptive statistics, including sample size, arithmetic mean, median, minimum, maximum, standard error (SE), and standard deviation (SD), were calculated by lake, sex, and maturity for each of the lethal and non-lethal Lake Chub survey and are presented in Appendix 7G (Table 7G-4 and Table 7G-5, respectively). The mean ±1 SD for each biological endpoint is provided in Sections 7.4.3 and 7.4.4.

Fish health indices, including Fulton's condition factor (K), LSI, and GSI, were calculated using both carcass weight (i.e., weight of fish with all organs and parasites removed) and total body weight. Indices were calculated as follows:

Condition Factor $K = (\frac{carcass\ weight}{fork\ length^3})\ x\ 100{,}000;$ [Equation 7-3]

Gonadosomatic index $GSI = \left(\frac{gonad\ weight}{carcass\ weight}\right) \times 100;$ [Equation 7-4]

Liversomatic Index $LSI = \left(\frac{liver\ weight}{carcass\ weight}\right) \times 100;$ [Equation 7-5]

where weight measurements are in grams and length is in millimeters.

7.2.3.5 Length-Frequency Distribution

Differences in the length-frequency distribution of Lake Chub among lakes were assessed using the non-parametric, two sample K-S test (Sokal and Rohlf 2012). The K-S test is best suited for testing differences in distributions based on continuous data, as it measures differences of the entire distribution, rather than other tests which are based on ranks. Fish data from the lethal survey and the non-lethal survey were used in the length-frequency analysis.

7.2.3.6 Lethal Survey Analyses

The lethal fish health survey considered the following effect indicators:

- survival as reflected by age and length-frequency distribution endpoints;
- energy use as reflected in relative gonad size, size at age, and fecundity endpoints; and,
- energy storage as reflected in condition, relative liver size, and relative egg size endpoints.

Survival is a measure of the difference in the mean age of all fish (separated by sex) between exposure and reference areas. A healthy fish population should exhibit variability in age.

Energy Use is a measure of the ability of the fish population to utilize resources in their environment to grow and reproduce. It is also an indicator as to whether a population is growing and reproducing normally and successfully.

Energy Storage is a measure of the current condition of the fish population. A healthy fish will demonstrate a greater body weight-to-length ratio and have a liver weight that is proportional to its body size and reproductive status.

Statistical procedures applied to the lethal fish health analyses are presented in Table 7-4. Additional supporting analyses that were added to the Snap Lake fish health program were size-at-maturity, stomach content analyses, and liver triglyceride and glycogen concentrations. Liver triglyceride and glycogen concentrations were protein-normalized prior to statistical analyses, by dividing triglyceride and glycogen concentrations (milligram per gram [mg/g]) by liver protein concentration (mg/g). Differences in age-at-maturity among lakes were not assessed statistically due to the use of length-based age assignments; therefore, size-at-maturity was examined visually by considering the GSI of all lethally sampled fish relative to fish length.

Table 7-4 Statistical Procedures Used in the Lethal Lake Chub Survey for Identifying Differences Between Snap Lake and the Reference Lakes

Effect Indicator	Endpoint	Dependent Variable (Y)	Covariate (X)	Statistical Procedure
Survival	Age	n/a	n/a	K-W
Growth (Survival)	Fork length	n/a	n/a	ANOVA
	Body weight	n/a	n/a	ANOVA or K-W
	Size-at-age	Carcass weight	Age	ANOVA or K-W
		Fork length	Age	ANOVA or K-W
	Length-frequency distribution	n/a	n/a	K-S test
Reproduction (Energy Use)	Relative gonad size	Gonad weight	Carcass weight	ANOVA and ANCOVA
		Gonad weight	Fork length	ANCOVA
	Relative fecundity	# eggs/female	Carcass weight	K-W
Condition (Energy Storage)	Condition	Total body weight	Fork length	ANCOVA
		Carcass weight	Fork length	ANOVA and ANCOVA
	Relative liver size	Liver weight	Carcass weight	ANOVA and ANCOVA
		Liver weight	Fork length	ANCOVA
	Egg size	Mean egg diameter	n/a	K-W

n/a = not applicable; K-W = Kruskal Wallis; ANOVA = Analysis of Variance; K-S test = 2-sample Kolmogorov-Smirnov test; ANCOVA = Analysis of Covariance; Carcass weight = measured carcass weight after removal of liver, gonads, stomach, intestines, and aging structures.

Analysis of Variance (ANOVA) and Analysis of Covariance (ANCOVA) were used to test for differences between Snap Lake and the reference lakes when data met parametric assumptions. The non-parametric test Kruskal-Wallis (K-W) was performed if neither the raw nor the $\log_{10}$ transformed data satisfied the normality and equality of variance requirements of parametric statistics. Alpha (α) and beta (β) were set equal at 0.1 for all statistical analyses (Environment Canada 2012), resulting in a statistical power (i.e., 1-β) of 90%.

ANOVA

If a significant difference was detected among all lakes following ANOVA (i.e., *P*<0.1), a paired contrast was performed to test for differences between Snap Lake and the pooled reference lakes (i.e., Northeast Lake and Lake 13 combined). To test the suitability of Lake 13 as a reference lake, paired contrasts comparing Lake 13 to Northeast Lake were also conducted. The magnitude of the difference between Snap Lake and the reference lakes for ANOVAs was calculated by expressing the difference as a percentage of the pooled mean of the two referenced lakes:

$$Magnitude\ Difference\ (\%) = \frac{(Exposure\ Mean - Pooled\ Reference\ Mean)}{Pooled\ Reference\ Mean} \times 100. \qquad \text{[Equation 7-6]}$$

The magnitude of the difference between reference lakes for ANOVAs was calculated as the relative percent difference:

$$Relative\ Percent\ Difference\ (\%) = \frac{(Reference\ 1\ Mean - Reference\ 2\ Mean)}{Pooled\ Reference\ Mean} \times 100. \qquad \text{[Equation 7-7]}$$

ANCOVA

An assumption of ANCOVA is that the slopes of the regression lines among treatment groups are equal; therefore, a test for homogeneity of slopes was conducted prior to performing ANCOVA analyses. If there was no significant interaction between sampling lakes and the covariate (i.e., assumption of homogeneity of slopes was satisfied), then an ANCOVA was performed and the adjusted means were calculated. If the slopes of the regression lines were found to be different, ANCOVA was conducted only if the coefficient of determination (R^2) from the test of slope interaction was large ($R^2 \geq 0.8$), and if the difference in the R^2 value from the full regression model (i.e., including the interaction term), and the partial regression model (i.e., not including the interaction term) from the ANCOVA was small (i.e., $\Delta R^2 \leq 0.02$) (Barrett et al. 2009). If there was a significant interaction, and the ΔR^2 greater than 0.02, the ANCOVA could not be performed and an ANOVA was run on the calculated index and the dependent variable.

If a significant difference was determined in the ANCOVA analyses, paired contrasts were performed to test for differences between Snap Lake and the pooled reference lakes (i.e., Northeast Lake and Lake 13 combined). Paired contrasts comparing Lake 13 to Northeast Lake were also conducted. The magnitude of the differences among lakes for ANCOVAs was calculated with least squared means (LSM):

$$Magnitude\ Difference\ (\%) = \frac{(Exposure\ LSM - Reference\ LSM)}{Reference\ LSM} \times 100. \qquad \text{[Equation 7-8]}$$

The magnitude of the difference between reference lakes for ANCOVAs was calculated as the relative percent difference:

$$Relative\ Percent\ Difference\ (\%) = \frac{(Reference\ 1\ LSM - Reference\ 2\ LSM)}{Pooled\ Reference\ LSM} \times 100. \qquad \text{[Equation 7-9]}$$

7.2.3.7 Non-Lethal Survey Analyses

Statistical procedures used for identifying differences between adult and juvenile Lake Chub from Snap Lake and the reference lakes are presented in Table 7-5. Condition factor (i.e., Equation 7-3) was calculated using total body weight for the non-lethal fish health survey.

Table 7-5 Statistical Procedures Used in the Non-Lethal Lake Chub Survey for Identifying Differences between Snap Lake and the Reference Lakes

Effect Indicator	Endpoint	Dependent Variable (Y)	Covariate (X)	Statistical Procedure
Survival	Length Frequency Distribution	n/a	n/a	K-S test
Growth (Energy Use)	Fork Length	n/a	n/a	ANOVA
	Total Body Weight	n/a	n/a	ANOVA
	Size-at-Age	n/a	n/a	K-W
Condition (Energy Storage)	Condition	Total Body Weight	Length	ANCOVA

n/a = not applicable; K-S test = 2-sample Kolmogorov-Smirnov test; ANOVA = Analysis of Variance; K-W = Kruskal Wallis; ANCOVA = Analysis of Covariance.

7.2.3.8 Power Analyses

Post-hoc power analyses were completed for all endpoints where significant differences were not detected to determine existing power for the analyses. Consistent with Chapter 1 of the MMER (Environment Canada 2012), the power to detect CES of 25% (for size-at-age, relative gonad size, relative liver size, gonad weight, liver weight, and liver triglyceride and glycogen concentrations) or 10% (for condition) were calculated. In addition, the sample size (n) required to detect the respective CES were calculated as an *a priori* consideration for future fish health programs. Power analyses were performed with the G*Power package (Faul et al. 2007).

7.1.1.1 Normal Range

Fish health endpoints were compared to the normal range (± 2 SD of the pooled reference lake mean) to determine whether any changes seen in Snap Lake during the 2012 fish health survey were beyond the range of variability present in the reference lakes.

7.3 QUALITY ASSURANCE AND QUALITY CONTROL

Quality assurance (QA) and quality control (QC) procedures were applied to all field sampling, laboratory analyses, data entry, data analyses and report preparation tasks to produce technically sound and scientifically defensible results.

7.3.1 Overview of Procedures

Field and lab equipment were calibrated throughout the field program as per specifications (i.e., daily or each use) and all samples were collected by experienced personnel. Samples were labelled, preserved, and shipped according to standard protocols. Specific work instructions outlining each field task in detail were provided to the field personnel by the task manager. Detailed field notes were recorded in waterproof field books and on pre-printed waterproof field data sheets in either pencil or indelible ink. Data sheets and sample labels were checked at the end of each field day for completeness and accuracy, and scanned into electronic copies at the completion of the field program. Chain-of-custody forms were used to track the shipment of samples.

Individual QA/QC procedures were undertaken by each laboratory performing specialty analyses for the 2012 fish health program:

- **Gonad Histology** - A subsample of the gonad histology data (10%) were randomly selected and re-analyzed by an independent histopathologist. The QA/QC results were discussed among the histopathologists and the gonad stages were adjusted according to consensus and reported in the laboratory report (Appendix 7B).
- **Aging** - A subsample of the aging structures (10%) was examined by a separate fish aging technician. Results of the age comparisons were provided in the analysis report (Appendix 7F).
- **Liver Glycogen and Triglyceride** - Each liver lipid assay was run at least twice with a portion of the samples from each run performed in duplicate. The same internal standard, Rainbow Trout liver homogenate, was run in parallel for every assay in triplicate. Assays where the internal standard varied more that 5% from previously determined norms were discarded and the assays were re-run. The inter-assay variability for each of the three end-points measured was calculated based on the coefficient of variation (CV) of the internal standard and these results are discussed in the laboratory report (Appendix 7F).
- **Stomach Contents** - A subsample of the stomach content samples (10%) was re-analyzed by the taxonomist. The results of the re-analysis were incorporated into the final taxonomy report (Appendix 7F).
- **Fecundity** - One out of every 10 fecundity samples was re-counted by a second, independent individual. If the re-count of the sample was within 10% of the initial count, the initial count was regarded as acceptable and no re-counts of the remaining samples were required. If the re-count was not within 10% of the initial count, the initial count was regarded as unacceptable and the remaining nine samples were re-counted. The QA/QC procedure was repeated until re-counts were within 10% of the previous count. The results of the fecundity analyses are presented in the fecundity report (Appendix 7H)

The appendices provide the final reported results following internal QA/QC procedures reported by responsible laboratories and subsequent QA/QC procedures implemented upon receipt of the data. Results were screened visually upon initial receipt and any unusual or illogical results were flagged and the laboratory was asked to confirm their accuracy.

Data entry QA/QC involved checking a minimum of 10% of the data for completeness, data entry errors, transcription errors on field sheets, and invalid or impossible data values. If an error was found, data underwent a zero tolerance QC check, where every datum was checked.

Results of statistical results analyses were independently reviewed by a senior biologist with appropriate technical qualifications. Tables containing data summaries and statistical results were reviewed, and values verified by a second, independent individual.

7.3.2 Summary of Results

In general, laboratory results passed the QA/QC review and were deemed valid. There was a minor discrepancy in gonad histology and one set of data, the age data, was deemed invalid and is not used in this report:

- **Gonad Histology** - Gonad histology results were inconsistent with field observations on a few occasions (e.g., sex and maturity identified in the field did not correspond with the histology); these samples were discussed with the laboratory, re-examined by the histopathologist, and confirmed correct in the final report (Appendix 7B).
- **Aging** – Age data were reviewed and deemed invalid on the basis of the following:
  - In the QA/QC review, age technicians differed in assessment of the age of fish with differences up to two years;
  - the maximum age of 15 years observed in the data exceeded the maximum age for Lake Chub reported in the literature (Brown 1969; McPhail and Lindsey 1970); and,
  - the range of age determinations was not consistent or correlative with the length-frequency distribution.

Mackay et al. (1990) indicate that aging structures from cyprinids such as Lake Chub may be particularly challenging, and that length-frequency data can be used to assign ages. The length-frequency distributions and the assigned age classes plotted relative to the total body weight-fork length relationship that were used to determine length-frequency assigned ages, are presented in Appendix 7F. Further consideration on length-frequency based age assignments is discussed in Section 7.5 (Uncertainty). Data analyses and further interpretation based on age relates to age assigned by length-frequency and not otolith-age data from the laboratory.

7.4 RESULTS

7.4.1 Fish Capture Data

A total of 1,095 fish were collected during the 2012 fish health program; 266 fish were collected from Snap Lake, 172 fish from Northeast Lake, 136 fish from Lake 13, and 521 fish from Downstream Lake 1 (Figures 7-1 to 7-4). Fish captured were Lake Chub, Burbot (*Lota lota*), Arctic Grayling (*Thymallus articus*), Lake Trout, Longnose Sucker (*Catostomus catostomus*), Ninespine Stickleback (*Pungitius pungitius*), Northern Pike (*Esox lucius*), Round Whitefish, and Slimy Sculpin (Table 7-6). Arctic Grayling and Lake Trout were only captured in Snap Lake and

Northeast Lake. Longnose Sucker were only captured in Snap Lake and Downstream Lake 1. Round Whitefish were only collected in Snap Lake. Slimy Sculpin were captured in each lake in very low numbers (i.e., ≤3 fish). Ninespine Stickleback were also collected in low numbers from Snap Lake, Northeast Lake, and Lake 13, but not Downstream Lake 1. This is the first incidence of a Ninespine Stickleback being captured in Snap Lake. The presence of one Ninespine Stickleback in the lake may be a result of transplantation from other regional waterbodies by waterfowl or fishing gear (i.e., eggs transferred on nets), rather than an indication of a new species present in the lake; the presence of Ninespine Stickleback in Snap Lake is under review. Fish capture data are presented in Appendix 7I.

A total of 972 Lake Chub were captured during the 2012 fish health program; 238 were collected from Snap Lake, 118 from Northeast Lake, 100 from Lake 13, and 516 from Downstream Lake 1 (Table 7-7). Four adult male and two adult female Lake Chub were retained and archived from Downstream Lake 1 for potential future fish tissue chemistry analyses. Lake Chub were collected in the greatest numbers by minnow traps in both Snap Lake and Downstream Lake 1, while electrofishing caught the most fish in Northeast Lake, and hoop netting caught the most fish in Lake 13 (Table 7-6). Overall, boat electrofishing resulted in the highest CPUE in Snap Lake and Northeast Lake (0.69 and 0.57 fish/100 seconds, respectively), while hoop netting was more productive in Lake 13 (0.52 fish/hour).

The relative fishing effort and fishing success for each gear type varied between lakes (Table 7-7). Fish captured among lakes by the different methods within each survey were examined visually (i.e., by boxplots) and appeared similar in length and weight with the exception of slightly longer fish being captured in Northeast Lake by boat electrofishing. Fish collected by different gear types were, therefore, pooled within a lake for subsequent analyses.

Table 7-6 Catch-per-Unit-Effort for Fish Captured in Snap Lake, Northeast Lake, Lake 13, and Downstream Lake 1 during the 2012 AEMP

Lake	Hoop Netting				Boat Electrofishing				Minnow Traps				All Effort Types			
	Effort (hours)	Species	# of Fish Captured	CPUE (# fish/hr)	Effort (seconds)	Species	# of Fish Captured	CPUE (# fish/100 sec)	Effort (hours)	Species	# of Fish Captured	CPUE (# fish/hr)	Effort (hours)	Species	# of Fish Captured	CPUE (# fish/hr)
Snap Lake	78.37	ARGR	0	-	3456	ARGR	4	0.12	1542.90	ARGR	0	-	1622.23	ARGR	4	0.00
		BURB	0	-		BURB	1	0.03		BURB	0	-		BURB	1	0.00
		LKCH	24	0.31		LKCH	24	0.69		LKCH	190	0.12		LKCH	238	0.15
		LKTR	0	-		LKTR	1	0.03		LKTR	0	-		LKTR	1	0.00
		LNSC	12	0.15		LNSC	4	0.12		LNSC	1	0.00		LNSC	17	0.01
		NNST	0	-		NNST	0	-		NNST	1	0.00		NNST	1	0.00
		NRPK	0	-		NRPK	0	-		NRPK	0	-		NRPK	0	-
		RNWH	1	0.01		RNWH	0	-		RNWH	0	-		RNWH	1	0.00
		SLSC	0	-		SLSC	3	0.09		SLSC	0	-		SLSC	3	0.00
		Total	**37**	**0.47**		**Total**	**37**	**1.07**		**Total**	**192**	**0.12**		**Total**	**266**	**0.16**
Northeast Lake	91.25	ARGR	0	-	19188	ARGR	2	0.01	265.90	ARGR	0	-	362.48	ARGR	2	0.01
		BURB	0	-		BURB	26	0.14		BURB	0	-		BURB	26	0.07
		LKCH	8	0.09		LKCH	109	0.57		LKCH	1	0.00		LKCH	118	0.33
		LKTR	2	0.02		LKTR	3	0.02		LKTR	0	-		LKTR	5	0.01
		LNSC	0	-		LNSC	0	-		LNSC	0	-		LNSC	0	-
		NNST	1	0.01		NNST	12	0.06		NNST	0	-		NNST	13	0.04
		NRPK	0	-		NRPK	6	0.03		NRPK	0	-		NRPK	6	0.02
		RNWH	0	-		RNWH	0	-		RNWH	0	-		RNWH	0	-
		SLSC	0	-		SLSC	2	0.01		SLSC	0	-		SLSC	2	0.01
		Total	**11**	**0.12**		**Total**	**160**	**0.83**		**Total**	**1**	**0.00**		**Total**	**172**	**0.47**
Lake 13	150.70	ARGR	0	-	3672	ARGR	0	-	1297.68	ARGR	0	-	1449.40	ARGR	0	-
		BURB	1	0.01		BURB	10	0.27		BURB	3	0.00		BURB	14	0.01
		LKCH	78	0.52		LKCH	14	0.38		LKCH	8	0.00		LKCH	100	0.07
		LKTR	0	-		LKTR	0	-		LKTR	0	-		LKTR	0	-
		LNSC	1	0.01		LNSC	0	-		LNSC	0	-		LNSC	1	0.00
		NNST	10	0.01		NNST	0	-		NNST	9	0.01		NNST	19	0.01
		NRPK	1	0.01		NRPK	0	-		NRPK	0			NRPK	1	0.00
		RNWH	0	-		RNWH	0	-		RNWH	0	-		RNWH	0	-
		SLSC	1	0.01		SLSC	0	-		SLSC	0	-		SLSC	1	0.00
		Total	**92**	**0.61**		**Total**	**24**	**0.65**		**Total**	**20**	**0.02**		**Total**	**136**	**0.09**
Downstream Lake 1	22.97	ARGR	0	-	n/a	ARGR	0	n/a	689.50	ARGR	0	-	712.47	ARGR	0	-
		BURB	1	0.04		BURB	0	n/a		BURB	0	-		BURB	1	0.00
		LKCH	103	4.48		LKCH	0	n/a		LKCH	413	0.60		LKCH	516	0.72
		LKTR	0	-		LKTR	0	n/a		LKTR	0	-		LKTR	0	-
		LNSC	1	0.04		LNSC	0	n/a		LNSC	2	0.00		LNSC	3	0.00
		NNST	0	-		NNST	0	n/a		NNST	0	-		NNST	0	-
		NRPK	0	-		NRPK	0	n/a		NRPK	0	-		NRPK	0	-
		RNWH	0	-		RNWH	0	n/a		RNWH	0	-		RNWH	0	-
		SLSC	0	-		SLSC	0	n/a		SLSC	1	0.00		SLSC	1	0.00
		Total	**105**	**4.57**		**Total**	**0**	n/a		**Total**	**416**	**0.60**		**Total**	**521**	**0.73**

= number; CPUE = Catch Per Unit Effort; s = seconds; hr = hour; n/a = not applicable (method not used); - = not calculated due to no catch; ARGR = Arctic Grayling; BURB = Burbot; LKCH = Lake Chub; LKTR = Lake Trout; LNSC = Longnose Sucker; NNST = Ninespine Stickleback; NRPK = Northern Pike; RNWH = Round Whitefish; SLSC = Slimy Sculpin.

Table 7-7 Number of Lake Chub Measured in the 2012 Snap Lake AEMP Fish Health Surveys by Gear Type

Fishing Method	Snap Lake		Northeast Lake		Lake 13		Downstream Lake 1	
	Lethal	Non-Lethal	Lethal	Non-Lethal	Lethal	Non-Lethal	Lethal	Non-Lethal
Boat Electrofishing	9	15	69	42	6	8	-	-
Hoop Net	24	-	1	5	45	33	-	103
Minnow Trap	56	132	1	-	7	1		264 (6)[(c)]
TOTAL	**89**	**147**	**71**	**47**	**58**	**42**	**6**	**367**
Combined Total	**236** [(a)]		**118**		**100**		**373** [(b)]	

(a) Two Lake Chub were collected during the field program from Snap Lake and were counted in the total catch but were not measured and not included in the lethal or non-lethal survey totals.

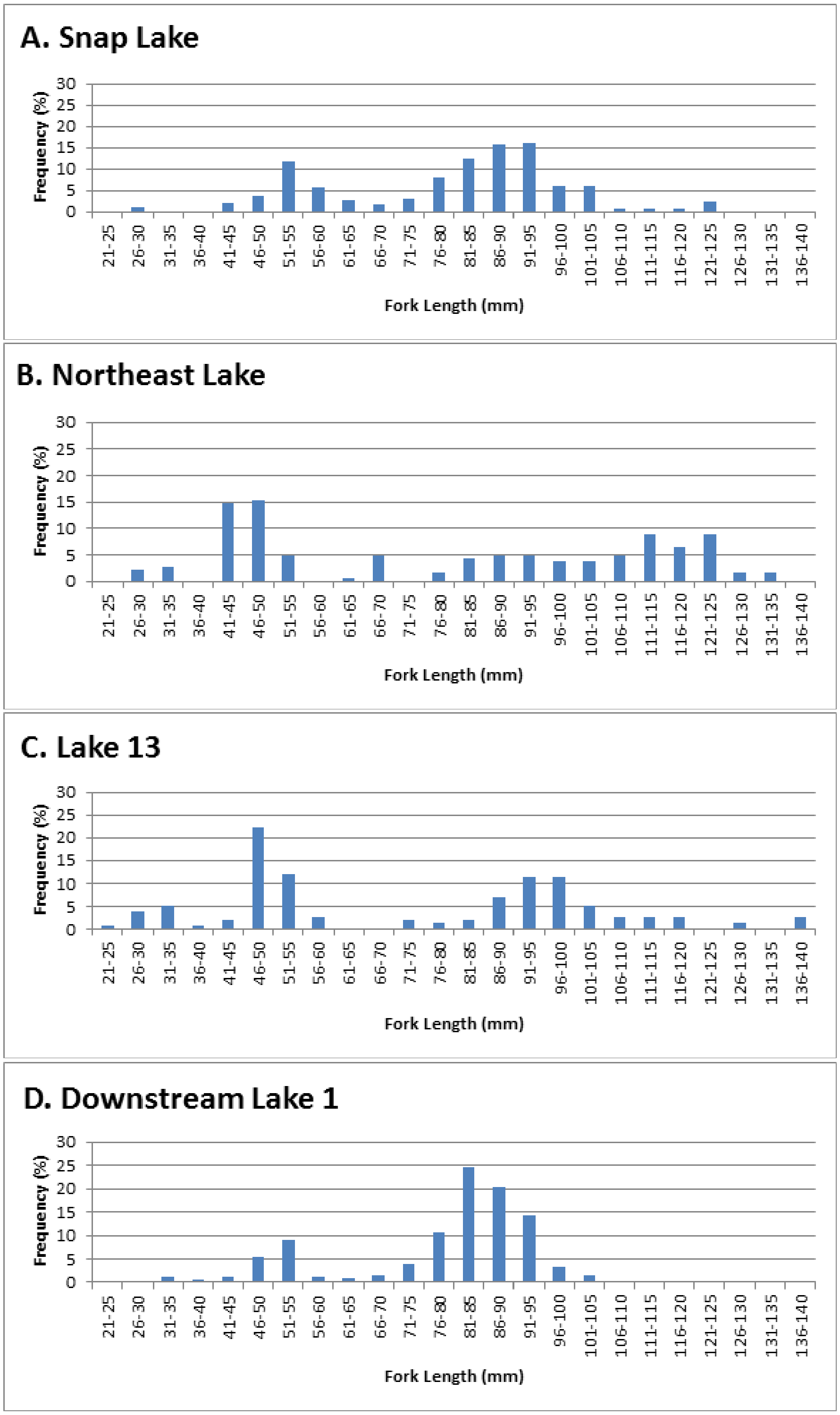
(b) Due to high catch numbers and limited time for processing, 149 Lake Chub from Downstream Lake 1 were counted in the total catch but were not measured and not included in the lethal or non-lethal survey totals.

(c) Six Lake Chub were captured during the non-lethal survey and were archived for future tissue analyses and are excluded from fish measurements.

7.4.2 Length Frequency Analysis

The length frequency distribution of Lake Chub collected from Snap Lake was significantly different than the pooled length frequency distribution from the reference lakes (P<0.001), and was also different from Northeast Lake (P=0.009), Lake 13 (P<0.001), and Downstream Lake 1 (P=0.016) when compared individually (Figure 7-1). The length frequency distribution of Lake Chub collected from Downstream Lake 1 was also significantly different from the pooled reference lakes (P<0.001), and was different from Northeast Lake (P=<0.001) and Lake 13 (P<0.001).

Figure 7-1 Length Frequency Distribution of Lake Chub from (A) Snap Lake, (B) Northeast Lake, (C) Lake 13, and (D) Downstream Lake 1



Note: Plots include lethal and non-lethal fish collections.

7.4.3 Lethal Survey

A total of 218 Lake Chub of a target 270 were used in the lethal fish health program during the 2012 AEMP from Snap Lake, Northeast Lake, and Lake 13 (Table 7-8). Downstream Lake 1 was not included in the lethal survey (see Section 7.4.4). Target sample sizes were not achieved due to unequal catch and limitations in sampling and processing time. In general, the target sample sizes were best achieved for Snap Lake. The raw data from the lethal Lake Chub fish health survey are presented in Appendix 7I. The gonad histology used in determining sex and maturity are presented in Appendix 7B.

Table 7-8 Total Number of Lake Chub Used in the Lethal Fish Health Survey from Snap Lake, Northeast Lake, and Lake 13 during the 2012 AEMP

Maturity	Sex	*Target n*	Achieved n		
			Snap Lake	Northeast Lake	Lake 13
Adult	Male	*30*	31	14	25
	Female	*30*	25	26	10
Juvenile	-	*30*	31	30	23
Unknown[(a)]	Unknown[(a)]	-	2	1	0
TOTAL		***90***	**89**	**71**	**58**

(a) Unknown = fish were lacking gonad histology data and could not be confirmed as male, female, or juvenile based on field observations or length; therefore, unknown fish were included in the length-frequency analyses and the pathology assessment, but were excluded from adult male, female, and juvenile fish health endpoint analyses.

n = sample size; - = sex was not considered in juvenile fish; Target N = desired sample size in each lake.

7.4.3.1 Pathology

External abnormalities were observed in 27 of the 218 fish. External abnormalities consisted of blindness, pale and/or frayed gills, skin aberrations, fin erosion, inflammation of the hindgut, and presence of external parasites (Table 7-9). The majority of observed external abnormalities were pale gills, which occurred in all three lakes (Appendix 7A).

Internal abnormalities were observed in 134 of the 218 fish. Internal abnormalities consisted of “fatty” liver, discolouration and/or nodules on the liver, enlarged spleen, parasites or enlarged gall bladder, and granular, mottled or swollen kidneys. The majority of the internal abnormalities were related to liver pathology; “fatty” livers were pale in colour, and were observed in all three lakes.

Table 7-9 External and Internal Abnormalities Observed in Lethally Sampled Lake Chub from Snap Lake, Northeast Lake, and Lake 13, 2012

Assessment Type	Category	Snap Lake	Northeast Lake	Lake 13
External	Body Deformities	0	0	1
	Eyes	1	0	0
	Gills	8	3	1
	Pseudobranchs	0	0	0
	Thymus	0	0	0
	Skin	0	2	1
	Fins	7	1	1
	Opercles	0	0	0
	Hindgut	1	0	1
Internal	Liver	39	19	21
	Spleen	0	0	2
	Gall Bladder	4	14	11
	Kidney	9	11	4
Total Number of Fish Surveyed		**89**	**71**	**58**

7.4.3.2 Parasites

Parasites were observed in fish from Snap Lake, Northeast Lake, and Lake 13 in the form of tapeworms and small white capsules or cysts (Table 7-10). Northeast Lake had the lowest incidence of parasitism, while Snap Lake and Lake 13 had similar proportions of infected Lake Chub. There were differences in incidence of tapeworm infection between sexes among the lakes; Northeast Lake had no female fish with tapeworms, while female fish from Lake 13 had the highest incidence of tapeworm infection. There was a similar proportion of fish with cysts among the lakes.

Table 7-10 Incidence of Parasites in Lake Chub in Snap Lake, Northeast Lake, and Lake 13, 2012

Sampling Area	Sex	Total Number of Fish Sampled	Number of Fish with Tapeworms		Number of Fish with Cysts		Severity					
							Low		Moderate		Severe	
			n	%	n	%	n	%	n	%	n	%
Snap Lake	Male	30	2	7	5	17	6	20	0	0	1	3
	Female	23	3	13	3	13	5	22	1	4	0	0
	Juvenile	31	4	13	12	39	15	48	1	3	0	0
	Total	**84[(a)]**	**9**	**11**	**20**	**24**	**26**	**31**	**2**	**2**	**1**	**1**

Table 7-10 Incidence of Parasites in Lake Chub in Snap Lake, Northeast Lake, and Lake 13, 2012

Sampling Area	Sex	Total Number of Fish Sampled	Number of Fish with Tapeworms		Number of Fish with Cysts		Severity					
							Low		Moderate		Severe	
			n	%	n	%	n	%	n	%	n	%
Northeast Lake	Male	14	1	7	3	21	4	29	0	0	0	0
	Female	26	0	0	8	31	8	31	0	0	0	0
	Juvenile	30	0	0	4	13	4	13	0	0	0	0
	Total	**70[(a)]**	**1**	**1**	**15**	**21**	**16**	**23**	**0**	**0**	**0**	**0**
Lake 13	Male	25	2	8	9	36	11	44	0	0	0	0
	Female	10	2	20	2	20	3	30	1	10	0	0
	Juvenile	23	1	4	5	22	6	26	0	0	0	0
	Total	**58**	**5**	**9**	**16**	**28**	**20**	**34**	**1**	**2**	**0**	**0**

(a) Fish that were classified as maturing or unknown sex were not included in the parasite incidence calculations.
n = sample size;% = percentage of total fish sampled with tapeworms;

7.4.3.3 Survival, Growth, Reproduction and Condition

Fish health endpoints related to survival (e.g., age), growth (e.g., size at age), reproduction (e.g., relative gonad size, relative fecundity), and condition (e.g., condition, relative liver size) are considered separately in the following sections.

Age

Adult Lake Chub from Snap Lake were significantly younger than adults from the reference lakes (Table 7-11); male adults were 12% and female adults were 21% younger than adults from the reference lakes (Table 7-12). Juvenile fish were defined as age one fish based on the length-frequency distribution; therefore, no comparison among lakes in juvenile age was performed. Northeast Lake males were 12% older than Lake 13; there was no difference in the ages of the reference lake females. In general, because the differences detected in Snap Lake were also found between reference lakes, the difference in Snap Lake is not thought to be biologically significant.

Size

Male and female adult Lake Chub were significantly shorter and lighter than adults from the reference lakes (Tables 7-11 and 7-12). Male Lake Chub were 10% shorter and 26% to 27% lighter than males from the reference lakes (Table 7-12), while females were 13% shorter and 30% to 31% lighter than females from the reference lakes. Juvenile fish were significantly longer (5%) and heavier (12% to 13%) in Snap Lake compared to juvenile fish from the reference lakes (Table 7-12). Adult male and juvenile fish in Northeast Lake were 14% heavier than in Lake 13; adult females were not significantly different in length or weight between reference lakes.

Condition

Lake Chub were in similar condition in Snap Lake and the reference lakes; there were no significant differences in adult male, adult female, or juvenile Lake Chub condition among the lakes (Table 7-11 and 7-12). There was sufficient power to detect the CES of 10% difference between Snap Lake and the reference lakes in adult condition and juvenile condition (as calculated by total weight) (Table 7-12).

Size-at-Age

There were inconsistent trends in size-at-age of adult male and female Lake Chub at Snap Lake relative to the reference lakes (Table 7-11). Adult male Lake Chub were significantly shorter (6%) and lighter (14%) at two years of age in Snap Lake compared to the reference lakes (Table 7-12). There were no three year-old male Lake Chub collected from Snap Lake. Adult female Lake Chub size-at-age was not significantly different among the lakes at age two or three, and there was sufficient statistical power to detect a CES of 25% between Snap Lake and the reference lakes. Juvenile fish, or Lake Chub at one year of age, from Snap Lake were significantly longer (8%) and heavier (21%) than juvenile Lake Chub from the reference lakes. The magnitudes of the differences in size-at-age among adult and juvenile Lake Chub are low and below the CES (i.e., less than 25%); therefore, the differences are not thought to be biologically significant.

Liver Size

There were no differences in adult and juvenile Lake Chub relative liver sizes between Snap Lake and the reference lakes (Table 7-11 and 7-12). There was sufficient power to detect a CES of 25% between Snap Lake and the reference lakes in relative liver size for female Lake Chub (87% to 93% power), and reasonable power for juvenile Lake Chub (81% power); however, there was insufficient statistical power to detect differences in relative liver size for adult male Lake Chub (37% power; Table 7-12).

Liver Glycogen and Triglyceride

Liver triglyceride concentrations were significantly lower in male and juvenile Lake Chub from Snap Lake relative to the reference lakes (Tables 7-11 and 7-12). Liver triglyceride concentrations were not different among lakes in female Lake Chub, but there was insufficient statistical power to detect a difference.

Liver glycogen concentrations were not different among lakes in either male or female Lake Chub, but there was insufficient power to detect a CES of 25% between Snap Lake and the reference lakes. Juvenile Lake Chub from Snap Lake had significantly lower liver glycogen concentrations relative to the reference lakes (Table 7-12); the reference lakes were also significant different from each other.

Table 7-11 Summary Statistics for the Lethal Survey of Lake Chub from Snap Lake, Northeast Lake, and Lake 13

Sex	Parameter	Snap Lake		Northeast Lake		Lake 13	
		n	Mean ± SD	n	Mean ± SD	n	Mean ± SD
Male	Age (y)	29	2 ± 0	13	2 ± 1	23	2 ± 0
	Fork Length (mm)	29	89 ± 7	13	101 ± 8	23	97 ± 8
	Total Body Weight (g)	29	6.78 ± 1.55	13	10.06 ± 2.32	23	8.80 ± 2.38
	Carcass Weight (g)	29	6.21 ± 1.49	12	9.16 ± 2.27	21	8.03 ± 2.14
	Liver Weight (g)	26	0.06 ± 0.02	13	0.11 ± 0.05	23	0.09 ± 0.05
	Gonad Weight (g)	29	0.07 ± 0.02	13	0.09 ± 0.04	23	0.08 ± 0.02
	GSI (carcass weight)	29	1.16 ± 0.33	12	0.92 ± 0.23	21	1.01 ± 0.24
	LSI (carcass weight)	26	1.02 ± 0.35	12	1.13 ± 0.36	21	1.04 ± 0.39
	Triglyceride (protein normalized)	5	0.010 ± 0.006	5	0.025 ± 0.007	5	0.043 ± 0.018
	Glycogen (protein normalized)	5	0.077 ± 0.104	5	0.034 ± 0.027	5	0.092 ± 0.112
	Condition (carcass weight)	29	0.86 ± 0.06	12	0.88 ± 0.10	21	0.87 ± 0.07
	Condition (total weight)	29	0.94 ± 0.07	13	0.97 ± 0.11	23	0.94 ± 0.08
Female	Age (y)	20	2 ± 0	26	3 ± 0	8	3 ± 1
	Fork Length (mm)	20	100 ± 14	26	115 ± 14	8	113 ± 19
	Total Body Weight (g)	20	9.93 ± 4.96	26	14.43 ± 4.53	8	14.09 ± 7.96
	Carcass Weight (g)	20	8.39 ± 4.11	26	11.90 ± 3.54	8	11.93 ± 6.47
	Liver Weight (g)	20	0.19 ± 0.11	25	0.34 ± 0.15	8	0.29 ± 0.22
	Gonad Weight (g)	19	0.47 ± 0.21	26	0.81 ± 0.64	8	0.67 ± 0.47
	GSI (carcass weight)	19	6.37 ± 3.35	26	6.37 ± 3.56	8	5.43 ± 2.42
	Fecundity	19	901 ± 343	22	118 ± 172	5	746 ± 750
	Egg Diameter (μm)	19	831 ± 167	22	1051 ± 383	5	1091 ± 277
	LSI (carcass weight)	20	2.25 ± 0.54	25	2.81 ± 0.80	8	2.16 ± 0.64
	Triglyceride (protein normalized)	5	0.008 ± 0.005	5	0.009 ± 0.010	5	0.012 ± 0.003
	Glycogen (protein normalized)	5	0.098 ± 0.0.113	5	0.038 ± 0.037	5	0.106 ± 0.069
	Condition (carcass weight)	20	0.78 ± 0.06	26	0.76 ± 0.05	8	0.76 ± 0.06
	Condition (total weight)	20	0.92 ± 0.08	26	0.92 ± 0.07	8	0.88 ± 0.09
Juvenile	Age (y)	27	1 ± 0	30	1 ± 0	22	1 ± 0
	Fork Length (mm)	27	54 ± 7	30	52 ± 11	22	51 ± 6
	Total Body Weight (g)	27	1.60 ± 0.79	30	1.51 ± 1.12	22	1.31 ± 0.42
	Carcass Weight (g)	27	1.39 ± 0.72	30	1.32 ± 0.99	22	1.14 ± 0.37
	Liver Weight (g)	26	0.02 ± 0.02	30	0.02 ± 0.02	20	0.02 ± 0.01
	LSI (carcass weight)	26	1.75 ± 0.88	30	1.74 ± 0.58	20	1.53 ± 0.49
	Triglyceride (protein normalized)	5	0.017 ± 0.007	5	0.031 ± 0.009	5	0.030 ± 0.015
	Glycogen (protein normalized)	5	0.038 ± 0.019	5	0.145 ± 0.049	5	0.058 ± 0.073
	Condition (carcass weight)	27	0.83 ±0.13	30	0.83 ± 0.10	22	0.87 ± 0.10
	Condition (total weight)	27	0.96 ± 0.17	30	0.95 ± 0.12	22	1.00 ± 0.13

Note: Lethal statistics were performed on adult “seasonally developed” male and female fish (not including “maturing” fish) that did not have tapeworms.

SD = standard deviation; n = number; g = grams; y = years; mm = millimetres; μm = micrometres; GSI = gonadosomatic index, LSI = liver somatic index.

Table 7-12 Statistical Comparison of Lethal Survey Parameters Measured in Lake Chub from Snap Lake, Northeast Lake, and Lake 13

Sex	Parameter	Statistical Test	Overall *P*-value	Exposure (Snap Lake) vs. Reference Comparisons		Reference (Northeast Lake vs. Lake 13) Comparisons		*Post Hoc* Power Analysis (Snap Lake vs. Reference)		*A Priori* Sample Size Required to Detect CES (Snap Lake vs. Reference)
				P	%	*P*	%	Actual Achieved Power	Power to Detect CES [(c)]	n [(c)]
Male	Age	K-W	***	**	-12	*	12	--	--	--
	Fork Length	ANOVA	***	***	-10	ns	n/a	--	--	--
	Total Weight	ANOVA^{log}	***	***	-27	*	14	--	--	--
	Carcass Weight	ANOVA^{log}	***	***	-26	ns	n/a	--	--	--
	Liver Weight	ANOVA^{log}	**	**	-36	*	27	--	--	--
	Gonad Weight	K-W	ns	n/a	n/a	n/a	n/a	51	80	51
	Relative gonad size (gonad weight against carcass weight)	ANCOVA	ns	n/a	n/a	n/a	n/a	34	98	24
	Relative gonad size (gonad weight against fork length)	RNS	RNS	RNS	RNS	RNS	RNS	RNS	RNS	RNS
	Relative liver size (liver weight against carcass weight)	ANCOVA^{log}	-	-	-	-	-	-	-	--
	Relative liver size (liver weight against fork length)	ANCOVA^{log}	ns	n/a	n/a	n/a	n/a	10	27↑/37↓	247↑149↓
	Liver Triglyceride (protein normalized)	ANOVA^{log}	**	***	-70	ns	n/a	--	--	--
	Liver Glycogen (protein normalized)	ANOVA^{log}	ns	n/a	n/a	n/a	n/a	10	12↑/14↓	432↑/260↓
	Condition (total weight against fork length)	ANCOVA [(a)]	ns	n/a	n/a	n/a	n/a	10	100	14
	Condition (carcass weight against fork length)	ANCOVA	ns	n/a	n/a	n/a	n/a	11	100	13
	Age-2 size (fork length) at age	K-W	*	*	-6	ns	n/a	--	--	--
	Age-2 size (carcass weight) at age	K-W	*	*	-14	ns	n/a	--	--	--
	Age-3 size (fork length) at age	K-W	ns	no data	no data	n/a	n/a	no data	no data	no data
	Age-3 size (carcass weight) at age	ANOVA	ns	no data	no data	n/a	n/a	no data	no data	no data
Female	Age	K-W	***	***	-21	ns	n/a	--	--	--
	Fork length	ANOVA	**	**	-13	ns	n/a	--	--	--
	Total weight	ANOVA^{log}	**	**	-31	ns	n/a	--	--	--
	Carcass weight	ANOVA^{log}	*	**	-30	ns	n/a	--	--	--
	Liver weight	ANOVA^{log}	**	*	-43	ns	n/a	--	--	--
	Gonad weight	ANOVA^{log}	*	*	-40	ns	n/a	--	--	--
	Relative gonad size gonad weight against carcass weight)	ANCOVA^{log}	ns	n/a	n/a	n/a	n/a	12	19↑/25↓	391↑/236↓
	Relative gonad size (gonad weight against fork length)	ANCOVA^{log}	ns	n/a	n/a	n/a	n/a	17	19↑/25↓	396↑/239↓
	Fecundity	K-W	**	ns	n/a	*	40	--	--	--
	Mean egg diameter	K-W	**	**	-23	ns	n/a	--	--	--
	Relative fecundity (number of eggs against carcass weight)	ANCOVA^{log}	***	*	11	*	7	--	--	--
	Relative liver size (liver weight against carcass weight)	ANCOVA	ns	n/a	n/a	n/a	n/a	27	93	23
	Relative liver size (liver weight against fork length)	ANCOVA	ns	n/a	n/a	n/a	n/a	25	87	28
	Liver Triglyceride (protein normalized)	ANOVA^{log}	ns	n/a	n/a	n/a	n/a	16	16↑/20↓	166↑/100↓
	Liver Glycogen (protein normalized)	ANOVA^{log}	ns	n/a	n/a	n/a	n/a	12	11↑/12↓	684↑/412↓
	Condition (total weight against fork length)	ANCOVA	ns	n/a	n/a	n/a	n/a	39	96	21
	Condition (carcass weight against fork length)	ANCOVA	ns	n/a	n/a	n/a	n/a	59	99	15
	Age-2 size (fork length) at age	ANOVA	*	ns	n/a	*	11	--	--	--
	Age-2 size (carcass weight) at age	ANOVA	*	ns	n/a	*	31	--	--	--
	Age-3 size (fork length) at age	K-W	ns	n/a	n/a	n/a	n/a	25	95	3
	Age-3 size (carcass weight) at age	ANOVA	*	*	16	ns	n/a	--	--	--
Juvenile	Fork length	K-W	**	**	5	ns	n/a	--	--	--
	Total weight	K-W	*	*	13	*	14	--	--	--
	Carcass weight	K-W	*	*	12	*	15	--	--	--
	Liver weight	K-W	ns	n/a	n/a	n/a	n/a	19	36	178
	Relative liver size (liver weight against carcass weight)	ANCOVA	ns	n/a	n/a	n/a	n/a	14	81	50
	Relative liver size (liver weight against fork length)	Variances not equal	-	-	-	-	-	-	-	-
	Liver Triglyceride (protein normalized)	ANOVA^{log}	*	*	-45	ns	n/a	--	--	--
	Liver Glycogen (protein normalized)	ANOVA^{log}	*	*	-62	*	-94	--	--	--
	Condition (total weight against fork length)	ANCOVA	ns	n/a	n/a	n/a	n/a	61	97	25
	Condition (carcass weight against fork length)	ANCOVA$^{log\ (b)}$	ns	n/a	n/a	n/a	n/a	33	44↑/50↓	146↑/120↓
	Age-1 size (fork length) at age	K-W	***	***	8	*	5	--	--	--
	Age-1 size (carcass weight) at age	ANOVA^{log}	***	**	21	**	20	--	--	--

Note:Lethal statistics were performed on adult “seasonally developed” male and female fish (not including “maturing” fish) that did not have tapeworms.

(a) Could proceed with ANCOVA despite significant interaction because the difference between the full regression equation R^2 (0.918) and the partial regression equation R^2 (0.909) was less than 0.02 (Barrett et al. 2009).

(b) Could proceed with ANCOVA despite significant interaction because the difference between the full regression equation R^2 (0.940) and the partial regression equation R^2 (0.932) was less than 0.02.

(c) The power and required sample size to detect a change equivalent to the critical effect size (25%, or 10% for condition endpoints) is presented; where the power to detect an increase or a decrease from the reference mean is different, both values are shown (↑ = power or required sample size to detect an increase, ↓ = power or required sample size to detect a decrease).

P* <0.10; ** *P* <0.01; * *P* <0.001,% = magnitude difference (Exposure vs. Reference) or relative percent difference (Reference Comparisons); CES = critical effect size as described in Section 7.2.3.8; ANOVA = Analysis of Variance; ANCOVA = Analysis of Covariance; *P* = probability; R^2 = coefficient of determination; K-W = Kruskal Wallis; n = sample size; ns = not significant; n/a = not applicable; RNS = regression not significant; - = significant interaction (*P*<0.05) or unequal variance and ANCOVA could not proceed; -- = power not calculated because significant differences detected between exposure and reference; no data = no age-3 male fish collected from Snap Lake.

Gonad Size

There were no significant differences in male or female relative gonad size among the lakes (Table 7-12). There was sufficient statistical power to detect a CES of 25% between Snap Lake and the reference lakes for relative gonad size in male Lake Chub (98% power); however, there was insufficient statistical power for relative gonad size in female fish (22% to 32% power). Female gonad size was significantly smaller in Snap Lake (40% magnitude difference) when not considered relative to body size (Tables 7-11 and 7-12).

Size-at-Maturity

Lake Chub populations in Snap Lake, Northeast Lake, and Lake 13 appear to mature at 75 to 85 mm in length. This is indicated by the fish length at which the gonads of the majority of adult fish begin to increase substantially in size (i.e., observed increase in GSI in Figure 7-2). On the basis of a visual assessment, there were no differences in size at maturity among lakes.

In 2012, there was also no evidence of skip-spawning (alternate-year spawning) in Lake Chub populations in Snap Lake or the reference lakes (Figure 7-2; Appendix 7B). Mature fish were developing normally for the season; no fish was identified through field measurements or histology as an adult skip-spawner.

Fecundity

Relative fecundity, or the number of eggs against carcass weight, was significantly greater (11%) in Snap Lake when compared to the reference lakes. The relative fecundity of Northeast Lake females was 7% higher than Lake 13. These statistical differences are not thought to be biologically significant because the magnitude differences are less than the respective CES (i.e., <25%).

Mean egg diameter was significantly smaller in Snap Lake (23%; Table 7-12). Mean egg diameter was not different between the reference lakes. The differences in the egg diameter between Snap Lake and the reference lakes are not considered biologically significant because the magnitude differences are less than the respective CES (i.e., less than 25%).

Stomach Contents

The major taxa present in the stomach and the percent composition of each taxon were determined for lethally sampled Lake Chub from Snap Lake, Northeast Lake, and Lake 13 that had 50% or greater stomach fullness at the time of sampling. There were a limited number of samples available for analysis from Lake 13 because most stomachs were, for unknown reasons, empty. Stomach content data were summarized for male, female, and juvenile fish in Table 7-13; raw stomach content data are presented in Appendix 7D.

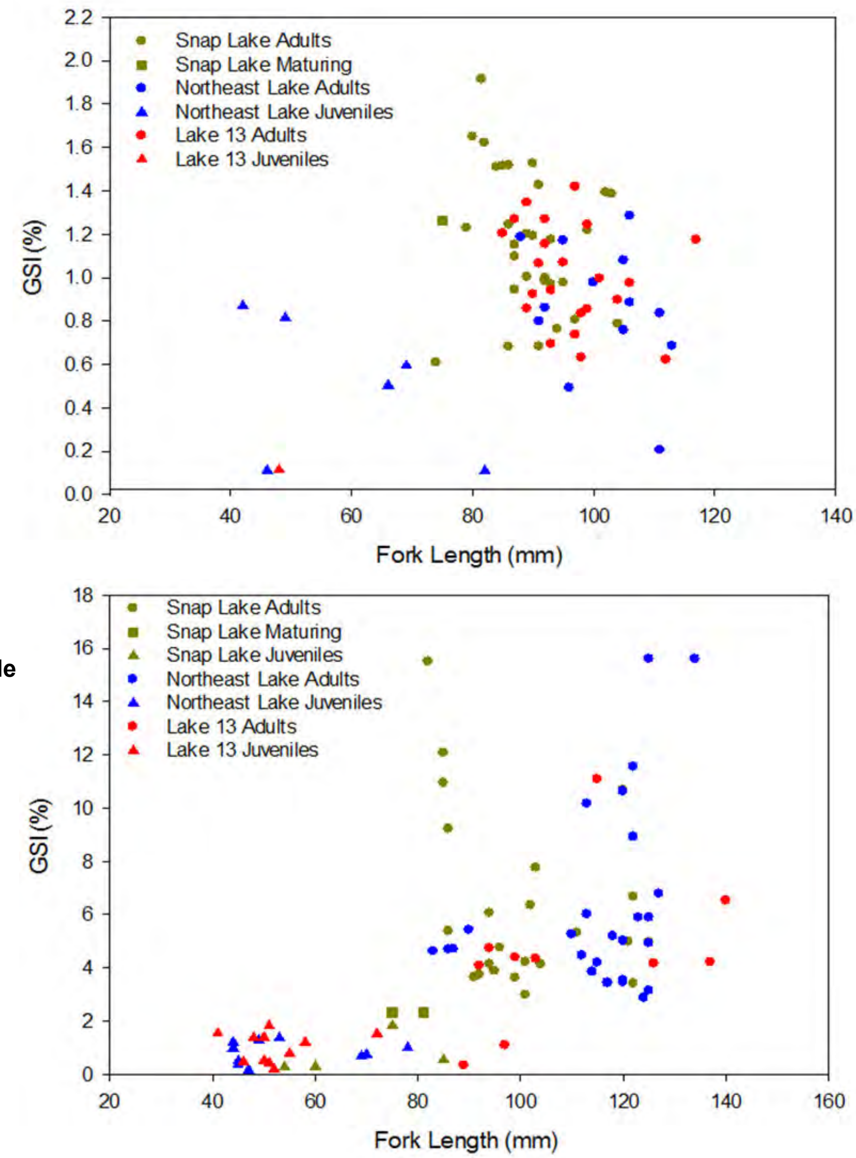
Terrestrial invertebrates were commonly found in the stomachs of Lake Chub from all lakes and were the dominant prey in stomachs of male, juvenile, and unknown fish (Table 7-13; Figure 7-3). Female stomachs had greater taxonomic diversity and included a higher proportion of

Gastropoda-Pelecypoda, Tricoptera, and Chironomidae than the male fish in both Snap Lake and Northeast Lake (Figure 7-3). Chironomidae, Coleoptera, and Trichoptera were absent from the fish collected in Lake 13, while adult Coleoptera and adult Arachnida were only found in stomach of fish from Lake 13. The dominant chironomids in Snap Lake were *Orthocladiinae* and *Tanypodinae,* while Northeast Lake was dominated by chironomid pupae and *Chironomidae* (Table 7-13; Figure 7-4).

Figure 7-2 Size at Maturity for (A) Male and (B) Female Lake Chub collected from Snap Lake, Northeast Lake, and Lake 13, 2012

(A) Male

(B) Female



Note: GSI = gonad somatic index; % = percent; mm = millimetre.

Table 7-13 Presence-Absence of Major Taxonomic Groups in Stomach Contents of Lethally Sampled Lake Chub captured in Snap Lake, Northeast Lake, and Lake 13.

Major Group	Snap Lake				Northeast Lake				Lake 13			
	M	F	J	U	M	F	J	U	M	F	J	U
Sample Size (n)	11	3	6	1	2	9	13	1	3	0	1	0
Terrestrial- adults of Chironomidae	X		X	X	X	X	X	X				
Terrestrial- adults of Trichoptera	X											
Terrestrial- adults of Diptera	X	X			X	X			X			
Terrestrial- adults of Tipulidae			X									
Terrestrial- adults of Coleoptera									X			
Terrestrial- Arachnida									X			
Gastropoda-Pelecypoda	X	X				X			X			
Cladocera			X			X	X	X			X	
Ostracoda							X					
Chironomidae - Pupa						X	X	X				
Chironomidae - Chiromoninae	X				X	X	X					
Chironomidae - Tanytarsini	X	X		X			X					
Chironomidae - Orthocladiinae	X	X	X			X	X					
Chironomidae - Tanypodinae	X	X	X				X					
Coleoptera						X	X	X				
Trichoptera		X			X	X						
Other	X	X	X			X	X					

M = male; F = female; J = juvenile; U = unknown (sex could not be determined in field and histology not available either through small gonad size or presence of parasite).

Figure 7-3 Composition of the Major Taxonomic Groups in Lake Chub Stomachs Captured in Snap Lake, Northeast Lake, and Lake 13, 2012

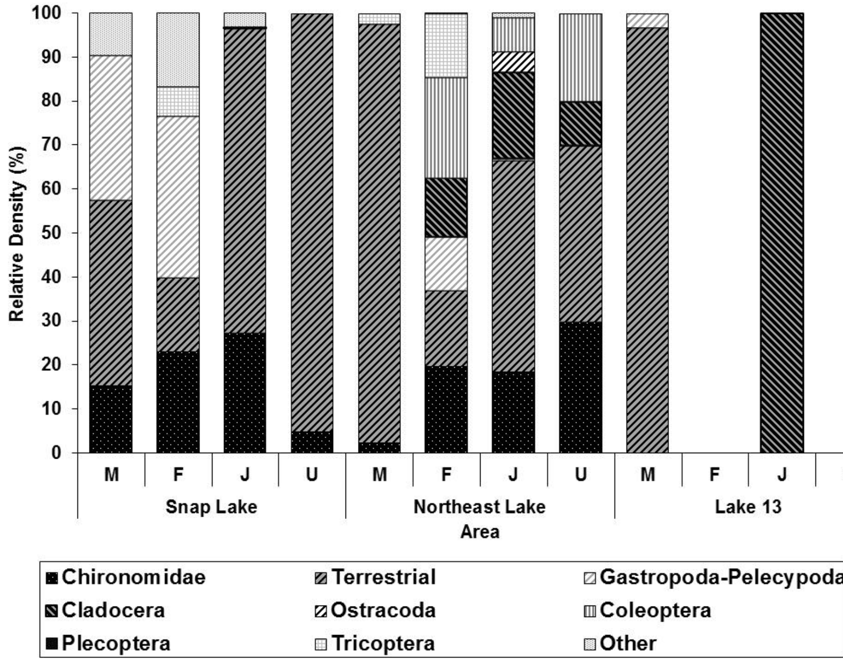
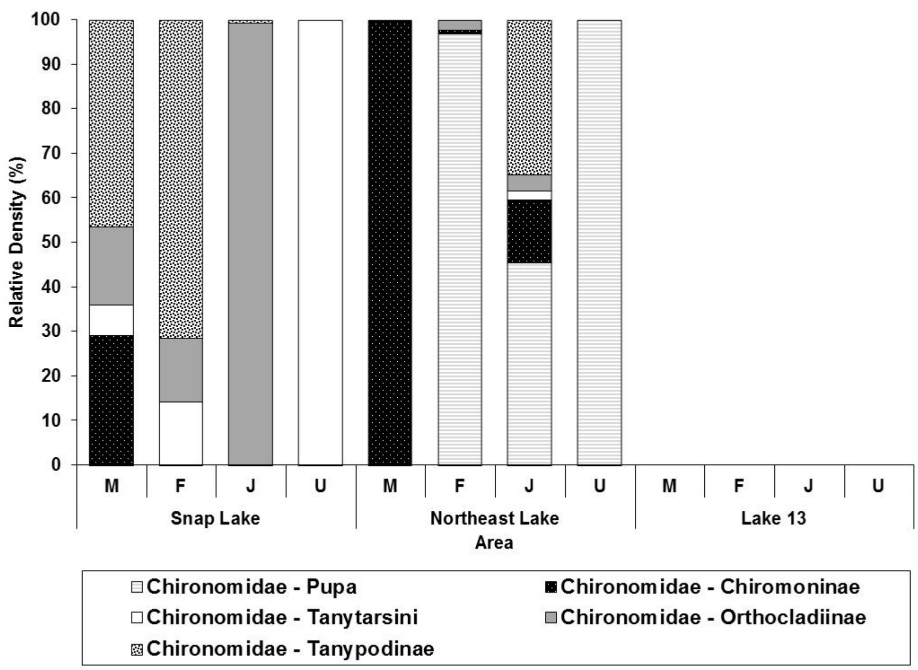


Figure 7-4 Chironomidae Composition in Lake Chub Stomachs Captured in Snap Lake, Northeast Lake, and Lake 13, 2012



7.4.4 Non-Lethal Survey

A total of 609 Lake Chub were included in the non-lethal fish health program from Snap Lake, Northeast Lake, Lake 13, and Downstream Lake 1 (Table 7-14). The non-lethal fish health raw data are presented in Appendix 7I.

Table 7-14 Total Number of Lake Chub Used in the Non-Lethal Fish Health Survey from Snap Lake, Northeast Lake, Lake 13, and Downstream Lake 1 during the 2012 AEMP

Target Species	Target N	Achieved N				
		Snap Lake	Northeast Lake	Lake 13	Downstream Lake 1	*Total N*
Lake Chub	100-400 non-YOY	147 [(a)]	47	42	373	***609***

(a) Includes two fish mortalities that were collected as part of the lethal program but were not included in the lethal survey.

n = sample size; YOY = Young-of-the-Year; Target N = desired sample size in each lake; therefore, total target sample size for the 2012 fish health program was 400-1200 non-YOY (i.e., juvenile and adult fish);

7.4.4.1 Pathological Assessment

External or visible abnormalities were observed in 15 of the 609 fish. External or visible abnormalities consisted of suspected parasites, skin deformities, and fin erosion or injury (Table 7-15). The majority of abnormalities were suspected tapeworms, which were observed in Snap Lake, Lake 13, and Downstream Lake 1. A skin abnormality was present on one fish which had a large black spot on the lateral surface; this fish was sent for histopathological examination. The black spot was determined to be a melanoma (Appendix 7B).

Table 7-15 External or Visible Abnormalities Observed in Non-Lethally Sampled Lake Chub from Snap Lake, Northeast Lake and Lake 13, 2012

Assessment Type	Category	Snap Lake	Northeast Lake	Lake 13	Downstream Lake 1
External	Skin	1	0	0	0
	Fins	3	0	0	3
	Parasites[(a)]	3	0	1	4
Total Number of Fish Surveyed		**147**	**47**	**42**	**373**
Total Number of Fish with Abnormalities		**7 (5%)**	**0 (0%)**	**1 (2%)**	**7 (2%)**

Note: Values in brackets represent percentage of total fish collected that presented abnormalities

(a) Large tapeworms visible by general body distension and visibility through ventral body cavity.

% = percent.

7.4.4.2 Survival, Growth and Condition

Size

Consistent with the lethal survey, non-lethally sampled adult Lake Chub were significantly shorter and lighter in Snap Lake relative to the reference lakes; juvenile Lake Chub were larger and heavier in Snap Lake compared to the reference lakes (Table 7-16 and 7-17). Adult and juvenile Lake Chub from Snap Lake were longer (3%) and heavier (6%) than fish from Downstream Lake 1 (Table 7-16 and 7-17). There were insufficient numbers of YOY collected from Snap Lake and Downstream Lake 1 to perform statistical analyses (Table 7-16).

Condition

Non-lethally sampled adult Lake Chub condition was significantly lower in Snap Lake relative to the reference lakes and Downstream Lake 1; however, the magnitude of difference was small and below that which would be considered biologically significant based on CES (i.e., 10%) (Table 7-16 and 7-17; Environment Canada 2012). Juvenile Lake Chub from Snap Lake had significantly greater condition than juvenile Lake Chub from the reference lakes, but were not different than their counterparts from Downstream Lake 1. Notably, Snap Lake juvenile condition was less different (i.e., smaller magnitude) than the juvenile condition in the reference lakes when compared to each other, and should thus be interpreted with caution (Table 7-16 and 7-17). There were insufficient numbers of YOY collected from Snap lake and Downstream Lake 1 to perform statistical analysis on YOY condition (Table 7-16).

Size-at-Age

Non-lethally sampled adult Lake Chub were not significantly longer or heavier at age two in Snap Lake relative to the reference lakes, but were significantly longer than adult Lake Chub from Downstream Lake 1 (Table 7-17). There was sufficient statistical power to detect a CES of 25% in size-at-age between Snap Lake and the reference lakes for adult Lake Chub (95% power). There were insufficient numbers of age three or age four adult Lake Chub collected from the study lakes to perform statistical analyses on size-at-age for three or four year-old Lake Chub.

Table 7-16 Summary Statistics for the Non-Lethal Survey of Lake Chub in Snap Lake, Downstream Lake 1, Northeast Lake, and Lake 13, 2012

Stage	Parameter	Exposure				Reference			
		Snap Lake		Downstream Lake 1		Northeast Lake		Lake 13	
		n	Mean ± SD	n	Mean ± SD	n	Mean ± SD	n	Mean ± SD
Adult	Fork length (mm)	114	88 ± 8	301	86 ± 7	19	98 ± 17	9	88 ± 10
	Total body weight (g)	115	6.58 ± 1.86	301	6.18 ± 1.47	19	9.63 ± 4.59	9	6.96 ± 2.43
	Condition (K) using total weight	114	0.94 ± 0.10	301	0.96 ± 0.11	19	0.96 ± 0.11	9	0.98 ± 0.07
Juvenile	Fork length (mm)	29	55 ± 6	69	51 ± 5	20	47 ± 6	18	49 ± 4
	Total body weight (g)	29	1.77 ± 0.54	69	1.39 ± 0.41	19	0.91 ± 0.29	18	1.26 ± 0.29
	Condition (K) using total weight	29	1.03 ± 0.13	69	1.03 ± 0.28	19	0.89 ± 0.25	18	1.04 ± 0.18
YOY	Fork length (mm)	3	27 ± 0	3	31 ± 1	8	30 ± 3	15	30 ± 4
	Total body weight (g)	3	0.60 ± 0.00	3	0.29 ± 0.05	8	0.29 ± 0.11	13	0.30 ± 0.06
	Condition (K) using total weight	3	3.05 ± 0.00	3	0.95 ± 0.19	8	1.04 ± 0.22	13	1.17 ± 0.40

n = sample size; SD = standard deviation; K = condition factor; mm = millimetre; g = gram; YOY = Young of the Year.

Table 7-17 Statistical Comparison of the Non-Lethal Survey Parameters Measured in Lake Chub from Snap Lake, Northeast Lake, and Lake 13

Stage	Parameter	Statistical Test	Overall P-value	Exposure (Snap Lake) vs. Reference Comparisons		Exposure (Snap Lake vs Downstream Lake 1) Comparison		Reference (Northeast Lake vs. Lake 13) Comparison		*Post Hoc* Power Analysis (Snap Lake vs Reference)		*A Priori* Sample Size Required to Detect CES (Snap Lake vs Reference)
				P	%	*P*	%	*P*	%	Actual Achieved Power	Power to Detect CES	n
Adult	Fork length (mm)	K-W	***	*	-7	**	3	*	1	--	--	--
Adult	Total body weight (g)	K-W	***	**	-25	*	6	ns	ns	--	--	--
Adult	Condition (total weight)	ANCOVA^{log}	**	**	-4	*	-1	ns	ns	--	--	--
Adult	Age-2 fork length (mm)	K-W	*	ns	ns	**	2	ns	ns	--	--	--
Adult	Age-2 total weight (g)	K-W	ns	ns	ns	*	-2	ns	ns	12	95	30
Juvenile	Fork length (mm)	K-W	***	***	15	**	8	*	-5	--	--	--
Juvenile	Total body weight (g)	K-W	***	***	64	***	27	***	-33	--	--	--
Juvenile	Condition (total weight)	ANCOVA^{log}	**	*	24	ns	ns	**	-35	--	--	--

*P <0.10; ** P <0.01; *** P <0.001, CES = critical effect size; P = probability; K-W = Kruskal Wallis; n = sample size;% = magnitude difference (Exposure vs. Reference) or relative percent difference (Reference Comparisons); mm = millimetre; g = gram; ns = not significant; n/a = not applicable; -- = power not calculated because significant differences detected between exposure and reference

7.4.5 Normal Range

All Lake Chub fish health endpoints in Snap Lake during the 2012 small bodied fish health program were within ± 2 SD normal range of the pooled reference lake mean (Table 7-18).

Table 7-18 Normal Range Summary of Fish Health Parameters Measured in the 2012 Lethal Lake Chub Fish Health Survey

Maturity/Sex	Endpoint	Normal Range [(a)] (Mean ± 2 SD)	Snap Lake Mean [(b)]
Adult	Fork length (mm)	63-127	(88) [86]
	Total body weight (g)	0.43-17.11	(6.58) [6.18]
	Condition (total weight)	0.77-1.17	(0.94) [0.96]
Adult (Male)	Fork length (mm)	83 - 115	89
	Age (years)	1 - 3	2
	Total body weight (g)	4.44 - 14.06	6.78
	Gonad weight (g)	0.02 - 0.15	0.07
	Liver weight (g)	0.00 - 0.20	0.06
	Carcass weight (g)	3.99 - 12.89	6.21
	GSI (%)	0.51 - 1.45	1.16
	LSI (%)	0.32 - 1.83	1.02
	Liver triglyceride (protein normalized)	0.002 – 0.066	0.010
	Liver glycogen (protein normalized)	0.000 – 0.229	0.077
	Condition (carcass weight)	0.72 - 1.03	0.86
	Condition (total weight)	0.77 - 1.13	0.94
Adult (Female)	Fork length	84 - 144	100
	Age	2 - 4	2
	Total body weight	3.58 - 25.12	9.93
	Gonad weight	0.00 - 1.97	0.47
	Fecundity	0 - 2201	901
	Mean egg size	502 - 1665	831
	Liver weight	0.00 - 0.66	0.19
	Carcass weight	3.34 - 20.48	8.39
	GSI	0.00 - 12.78	6.37
	LSI	1.04 - 4.27	2.25
	Liver triglyceride (protein normalized)	0.000 – 0.024	0.008
	Liver glycogen (protein normalized)	0.000 – 0.198	0.098
	Condition (carcass weight)	0.66 - 0.86	0.78
	Condition (total weight)	0.76 - 1.05	0.92
Juvenile	Fork length	33 - 70	54 (55) [51]
	Age	0 - 2	1
	Total body weight	0.00 - 3.20	1.60 (1.77) [1.39]
	Liver weight	0.00 - 0.05	0.02
	Carcass weight	0.00 - 2.83	1.39
	LSI	0.55 - 2.76	1.75
	Liver triglyceride (protein normalized)	0.009 – 0.053	0.017
	Liver glycogen (protein normalized)	0.000 – 0.251	0.038
	Condition (carcass weight)	0.64 - 1.05	0.83 (1.03) [1.03]
	Condition (total weight)	0.72 - 1.22	0.96

(a) Normal Range is the pooled reference mean (Northeast Lake + Lake 13) ± two standard deviations; <0 = lower end of normal range was less than zero;

(b) Values in round parentheses represent mean value from the non-lethal survey; adult fish in the non-lethal survey are presented separately due to lack of sex determination; values in square parenthesis represent mean value from the non-lethal Downstream Lake 1 survey.

LSI = liver somatic index; GSI = gonadosomatic index; SD = standard deviation;% = percent; g = gram; mm = millimetre.

7.4.6 Tissue Chemistry

The concentrations of chemicals measured in Lake Chub carcasses were not statistically different between Snap Lake and the reference lakes, with the exception of strontium and thallium (36% and 110% magnitude difference, respectively). The difference in thallium concentration between Snap Lake and the reference lakes was greater than the normal range (i.e., pooled reference mean ± 2SD), but not for strontium.

The range of water strontium concentrations measured in Snap Lake in 2012 was 500-750 microgram per litre (µg/L) (at a water hardness of 25.6 to 186 milligram per litre (mg/L); Section 3). These concentrations are orders of magnitude below concentrations that could adversely affect fish or other aquatic fauna (Pacholski 2009; Golder 2011b, 2013; Nautilus 2013). While Lake Chub carcass strontium concentrations were statistically greater in fish from Snap Lake than the reference lakes, reflective of different water concentrations between the lakes, there was no indication of impaired fish health or obvious pathology (Section 9).

Thallium water concentrations measured in Snap Lake and in the reference lakes in 2012 were below detection limits (i.e., maximum less than 0.01 µg/L). The source of elevated thallium concentrations Snap Lake Lake Chub carcasses is unknown; however, again, there was no indication of impaired fish health or obvious pathology (Section 9).

7.5 UNCERTAINTY

Uncertainty around two fish health endpoints exists in the 2012 fish health study, specifically fish age and fish size. The consequences of this uncertainty on the study results is discussed below.

As described in Section 7.3.2, ototlith age data were inconsistent with maximum Lake Chub ages reported in the literature, and also inconsistent with fish health endpoints included in the present study (i.e., fish length, weight and state of maturity as indicated by gonad histology were inconsistent with otolith age). As such, length frequency distributions were used to assign age. This introduces uncertainty in the analyses of the endpoints age, size-at-age, and age of maturity among lakes.

The relative fishing effort and fishing success of gear types varied among lakes, as described in Section 7.4.1. Fishing methods can select for certain sizes or sexes of fish (i.e., gear-selectivity), and the potential for gear-selectivity exists in the 2012 Lake Chub fish health survey. The differences in fishing success of the various gear types in the reference lakes (i.e., electrofishing and hoop nets) versus Snap Lake (i.e., minnow traps) could have introduced a bias in the size of Lake Chub collected among the lakes. However, this uncertainty may be minimal because fish sizes are consistent with previous programs, and all size classes were collected and are represented in 2012 in all lakes.Future programs will endeavor to improve, to the extent possible, consistency in fishing efforts by each gear types across the study lakes to reduce potential gear bias among lakes.

7.6 SUMMARY

Fish health indices such as condition, LSI, and GSI were not different between Snap Lake and the reference lakes in the 2012 fish health survey. These indices, or "effect endpoints" as defined in Chapter 8 of the MMER (Environment Canada 2012), carry high relative weight in determining whether changes to fish health are occurring. Additional fish health endpoints considered as supporting evidence, or "support endpoints", which were significantly different in Lake Chub from Snap Lake in 2012 were fork length, body weight, and fecundity. While these support endpoints were significantly different in Snap Lake relative to the reference lakes, they were all within normal ranges and were not of a magnitude that would be considered biologically significant (i.e., magnitude of difference less than 25%). There is no evidence to suggest fish health in Snap Lake has been affected as a result of changes in water or sediment quality to an extent beyond that which can be attributed to natural variation.

Downstream Lake 1 is located immediately downstream of Snap Lake, and has been influenced by the Mine, as indicated by changes in water quality (Section 3). Lake Chub collected from Downstream Lake 1 during the 2012 Lake Chub program were similar sized to those observed in Snap Lake and Lake 13. The presence of smaller-sized adult Lake Chub in Snap Lake and Downstream Lake 1 in the 2012 program is reflected in the length-frequency distribution; the mode for adult fish (i.e., the peak in distribution as shown in Figure 7-1) is more pronounced and centred around smaller fish in both Snap Lake and Downstream Lake 1 versus either Northeast Lake or Lake 13, where the adult fish mode has greater spread and is more inclusive of larger adults. All size classes were present in Snap Lake during the 2012 fish health survey.

The smaller size of adult Lake Chub in Snap Lake relative to the reference lakes is consistent with data collected in 2009 (Table 7-19). The mean size of Snap Lake fish is also consistent with measurements of fish in 2006, suggesting size of fish in Snap Lake has not changed over time (De Beers 2007). While liver weight was significantly lower in male and female Lake Chub in Snap Lake in 2012, there were no differences in LSI in female fish and insufficient power to detect differences in LSI in male fish; therefore, female liver size is not different among lakes when the size of the fish is taken into consideration. There was a significant decrease in liver triglyceride concentrations in male fish in Snap Lake, with a large magnitude of difference; however, mean liver triglyceride and glycogen concentrations in male fish were within ranges for the reference lakes. The lower liver weights in Lake Chub from Snap Lake, when considered alongside a lack of differences in fish condition or LSI among lakes, do not indicate impairment of energy storage or energy use in Lake Chub from Snap Lake.

Table 7-19 Summary of the Main Differences between Fish Health Parameters of Lake Chub collected from Snap Lake Relative To Fish from the Reference Lakes, 2012

Maturity/Sex	Endpoints >25% Different in Snap Lake versus the Reference Lakes in 2012	Was the Endpoint Mean Value Outside of Normal Range in 2012?	Was this Difference Seen in 2009?[(a)]
Adult Male	Total weight <in Snap Lake, magnitude of difference was -27%	No	Yes, magnitude of difference was - 24%
	Carcass weight <in Snap Lake, magnitude of difference was -26%	No	Yes, magnitude of difference was -19%
	Liver weight <Snap Lake, magnitude of difference was -36%	No	Yes; magnitude of difference was -37%
	Liver triglyceride <Snap Lake, magnitude of difference was -70%	No	n/a
Adult Female	Total weight <in Snap Lake, magnitude of difference was -31%	No	No, insufficient statistical power
	Carcass weight <in Snap Lake, magnitude of difference was -30%	No	No, insufficient statistical power
	Liver weight <Snap Lake, magnitude of difference was -43%	No	Not statistically tested, magnitude of difference was -27% [(b)]
	Gonad weight <Snap Lake, magnitude of difference was -40%	No	No, insufficient statistical power
Juvenile	Liver triglyceride <Snap Lake, magnitude of difference was -45%	No	n/a
	Liver glycogen <Snap Lake, magnitude of difference was -62%	No	n/a

Note: n/a = liver triglyceride and glycogen concentrations were not measured in 2009.

(a) Reported by De Beers (2010).

(b) = statistical test was not performed in 2009, magnitude of difference calculated based on means presented in Table 7-22 from De Beers (2010);.

>= greater than; <= less than;% = percent.

There were no trends in water temperature in Snap Lake relative to the reference lakes that would explain the differences observed in fish health endpoints (see Supporting Environmental Variables, Section 2). The differences observed between the Snap Lake and reference lakes fish populations, as presented in Table 7-19, could be due to a number of factors, either singly or in combination:

- Lake Chub had different food items in their stomachs at the time of capture among the study lakes. In particular, Snap Lake and Northeast Lake adult male and juvenile Lake Chub had notably diverse composition of stomach contents. While the sample sizes were limited in Lake 13 and unequal proportions of juvenile and adult fish were represented among the lakes, differences in diet may partially explain different liver triglyceride and glycogen concentrations in Lake Chub from Snap Lake.
- The relative fishing effort and fishing success of different gear types varied among the study lakes. Alternate fishing methods may introduce gear-selectivity for certain sizes of fish. However, it is unlikely gear selectivity influenced fish health endpoints because fish sizes in 2012 were consistent with previous programs, and all size classes are represented in 2012 in all lakes.

7.7 CONCLUSIONS

Lake Chub health is not being impacted in Snap Lake or Downstream Lake 1 due to changes in water or sediment quality to a degree greater than that seen by natural variability. While there were significant differences in Lake Chub health endpoints in Snap Lake relative to the reference lakes during the 2012 small bodied fish health program, all were within reference lake ranges, and many were of a magnitude of difference that is not considered biologically significant. Fish health indicators measured in Downstream Lake 1 were consistent with those measured in Snap Lake, and were within the ranges of the reference lakes.

7.7.1 Key Question 1: Is fish health affected by changes in water and sediment quality in Snap Lake?

It is unlikely that fish health has been affected by the changes in water and sediment quality in Snap Lake. Snap Lake fish health endpoints were within the range of the pooled reference lakes during the 2012 program, and conclusions were consistent with past programs (De Beers 2010, 2012). The fish health endpoints that were statistically different between Snap Lake and the reference lakes in 2012 were fish age, size, liver weight, gonad weight, liver glycogen and triglyceride concentrations, fecundity, and egg diameter; however, the magnitude of difference in the majority of endpoints was below that which would be considered biologically significant.

7.7.2 Key Question 2: Are changes observed in fish health greater than those predicted in the EAR?

Changes to fish health measured in 2012 are not greater than changes predicted in the EAR. The EAR predicted that chemicals of potential concern (i.e., particularly TDS, hexavalent chromium, and indirect effects via increased primary production) in water and sediment could have a negative effect on fish health, but that the magnitude of this effect would be negligible. No changes to fish reproduction were predicted. There is no direct evidence any of the differences measured in 2012 have affected the ability of fish to survive, grow, or reproduce in Snap Lake.

7.8 RECOMMENDATIONS

The following recommendations are provided for consideration in the future Snap Lake fish health AEMP program:

- **Downstream Lake Special Study** – The size of fish from Downstream Lake 1 is consistent with fish from Snap Lake. A future fish health program inclusive of Downstream Lake 1 is proposed to monitor fish health in the downstream lakes and to assess the extent of Mine influence on fish populations in the downstream waterbodies. It is recommended that a fish population survey be performed in Downstream Lake 1 prior to the next fish health program to determine whether similar fish populations and species are present and provide baseline information on lake ecology.

- **Fish Age** – Given the lack of reliable aging data from laboratory analyses, it is recommended that laboratory aging be excluded from future fish health programs until and unless an age validation study for Lake Chub is completed. It is recommended that the full non-lethal sample size (i.e., up to 400 fish, inclusive of fish size measurements) continue to be targeted to allow for age-determination based on length-frequency distributions.

- **Temporal trends and Age data** - In subsequent re-evaluations of the fish health component, otolith age data from past reports should be converted to the assigned age used in the 2012 AEMP such that comparisons of size at age and between adults and juveniles are consistent.

- **Normal Range** – Future studies should review the normal range of fish endpoints within Snap Lake main basin over time and between pooled references lakes. An assessment of the sensitivity of the normal range comparisons versus the CES comparisons towards determining the significance of an effect to fish health should be undertaken for the Snap Lake Mine.

- **Stomach Contents** – Stomach content analyses allow for consideration of potential differences in diet among lakes when interpreting Lake Chub health endpoints such as condition and relative liver size. Further, stomach content analyses allow integration of conclusions between the benthic invertebrate survey and potential consequences to fish health in Snap Lake. It is only possible to predict what changes in the benthic invertebrate community might have on fish species if the importance of individual invertebrate species, genera, or families as a prey item for fish is understood. It is recommended fish stomach content analyses be continued in future fish health programs.

- **Liver Triglyceride and Glycogen Concentrations** – Liver triglyceride and glycogen endpoints offer valuable supporting information when interpreting differences in fish condition, relative liver size, and reproduction. The liver triglyceride and glycogen endpoints were originally added due to increasing concentrations of nutrients and TDS and possible indicators of nutrient enrichment in Snap Lake, and it is recommended to continue their inclusion in future fish health programs. It is recommended to increase the number of liver samples analyzed from each lake to increase statistical power.

- **Gear-Selectivity** – In an effort to determine whether size bias as a result of gear selectivity is occurring in the study lakes, it is recommended that similar levels of effort by each fishing method be expended in each lake in future programs, regardless of initial fishing successes. While it is likely that differences in the success of various gear types will remain among lakes, a more balanced fishing effort with each gear type is required to determine whether a gear bias exists for size or sex of fish collected among the lakes.

- **Spawning Proportion and Sex Ratios** – Further to expending comparable amounts of time on the various fishing methods among lakes, it is recommended that the proportion of fish captured that are in spawning condition by each fishing method be calculated to examine whether reproductive status is comparable among fishing methods and among areas. Fish that will spawn will be defined as fish that are found to be pre-spawning, ripe, or spent. It is also recommended that the sex ratio of adult fish from Snap Lake and the reference lakes be examined in future programs to determine whether a different number of males and females are caught by different gear types in different lakes. If the sex ratio is not influenced by fishing method, sex ratios should be examined to determine whether differences exist in sex ratios among lakes or whether potential activities of males and females differ at the time of capture (e.g., spawning versus feeding).

7.9 REFERENCES

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8 FISH COMMUNITY

Specific Water Licence conditions applying to the fish community component of the Aquatic Effects Monitoring Program (AEMP) for the Snap Lake Mine in Water Licence MV2011L2-0004 [Part G, Schedule 6, Item 1a (iv) and 1d (MVLWB 2012)] are:

a) Monitoring for the purpose of measuring Project-related effects on the following components of the Receiving Environment:

iv. fish population, recruitment, and year-class strength and community composition;

d) Procedures to minimize the impacts of the AEMP on fish populations and fish habitat.

The fish community component of the Snap Lake AEMP is conducted every three to five years; it was last conducted in 2009, and will next be conducted in July 2013 and reported in the 2013 Annual Report.

9 FISH TISSUE CHEMISTRY

9.1 INTRODUCTION

In 2012, De Beers Canada Inc. (De Beers) implemented the field component of the Snap Lake Mine (Mine) Aquatic Effects Monitoring Program (AEMP), as required by the Type A Water Licence MV2011L2-0004 (MVLWB 2012). The scope of the AEMP is based on the study design document submitted to the Mackenzie Valley Land and Water Board (MVLWB) in June 2005, which was approved with conditions in July 2005. This section presents the results of the first small-bodied fish tissue chemistry survey conducted under the Mine's AEMP in 2012.

9.1.1 Background

Lake Trout (*Salvelinus namaycush*) and Round Whitefish (*Prosopium cylindraceum*) were included in a large-bodied fish survey in 1999, 2004, and 2009 to collect baseline data and document fish tissue chemistry in Snap Lake (De Beers 2002, 2005, 2010). No patterns in large-bodied fish tissue chemistry have been observed across the years (De Beers 2012a). The AEMP Design Plan was updated in 2012 (herein referred to as the 2013 AEMP Design Plan; De Beers 2012b), and a small-bodied fish survey using Lake Chub (*Couesius plumbeus*) was added to the fish tissue chemistry program. Small-bodied fish were added to the tissue chemistry program to provide an early indicator of potential changes in fish tissue chemistry and to support potential effects observed during the fish health study (De Beers 2012b). A second reference lake was also proposed in the 2013 AEMP Design Plan (De Beers 2012b). Accordingly, provisional reference Lake 13 (Lake 13) was added to the small-bodied fish tissue chemistry survey in 2012.

9.1.2 Objective

The objective of the Lake Chub tissue chemistry survey is to determine whether treated effluent discharged from the Mine is having an effect on fish tissue chemistry. Specific Water Licence conditions applying to the fish tissue chemistry component of the AEMP for the Mine in Water Licence MV2011L2-0004 [Part G, Schedule 6, Item 1a (iii) and 1(d) of MVLWB (2012)] are:

a) Monitoring for the purpose of measuring Project-related effects on the following:
- *v. contaminant levels in fish flesh due to changes in Water quality in Snap Lake and/or the NE Lake;*

d) Procedures to minimize the impacts of the AEMP on fish populations and fish habitat.

The fish tissue survey was designed to meet the above conditions by answering the following key question:

1. Are tissue metal concentrations in fish from Snap Lake increasing relative to reference lakes?

9.2 METHODS

9.2.1 Fish Collection and Laboratory Analyses

Fish were collected from one exposure lake (Snap Lake) and two reference lakes (Northeast Lake and Lake 13), as per the methods outlined in Section 7.2. The carcasses of eight Lake Chub collected from each of the three lakes were submitted for tissue chemistry analyses. Carcasses consisted of flesh and bone, but not viscera, liver or gonad tissues. The samples were fish used in the lethal assessment (Section 7). Therefore, the gonad, stomach and liver of each fish were not included as they were required for separate analyses for the fish health survey (Section 7.2). Each sample was selected based on having the same sex ratio (four males and four females from each lake) and size class (largest size class selected to meet the minimum sample volume requirement of 5 grams wet weight [g ww]) from each lake. The sex, length, and weight of the fish comprising these samples are provided in Table 9-1. Samples were analyzed by ALS Canada Ltd. (ALS; Burnaby, British Columbia) for metals[10] and lipid concentrations as listed in the 2013 AEMP Design Plan (De Beers 2012b; Table 9-2).

Table 9-1 Lake Chub Samples Used in the Fish Tissue Chemistry Survey, 2012 AEMP

Lake	Sex	Fork Length (mm)	Total Length (mm)	Carcass Weight (g ww)	Total Body Weight (g ww)
Lake 13	Female	126	136	15.23	17.87
		97	104	7.35	10.44
		140	150	23.85	29.28
		137	146	17.89	20.42
	Male	108	119	-	12.50
		116	125	-	13.79
		112	120	13.05	14.02
		117	122	13.37	14.76
Northeast Lake	Female	127	136	16.22	19.99
		134	143	18.45	23.74
		125	134	14.95	19.98
		125	137	15.53	17.73
	Male	106	121	12.88	14.29
		111	120	12.18	16.07
		105	115	11.13	12.23
		105	115	11.26	12.19
Snap Lake	Female	125	139	17.06	20.09
		120	129	13.13	16.52
		121	136	15.25	18.95
		122	137	15.66	17.96

[10] In the 2012 AEMP Annual Report, “metal” includes metalloids such as arsenic, and non-metals such as selenium.

Table 9-1 Lake Chub Samples Used in the Fish Tissue Chemistry Survey, 2012 AEMP

Lake	Sex	Fork Length (mm)	Total Length (mm)	Carcass Weight (g ww)	Total Body Weight (g ww)
Snap Lake	Male	104	110	9.28	9.84
		102	108	8.64	9.36
		95	104	8.22	8.87
		103	113	10.01	12.75

"-" = carcass weight not recorded in field; mm = millimetres; g ww = grams wet weight.

Table 9-2 Parameters Analyzed in Lake Chub Tissue Samples from Lake 13, Northeast Lake, and Snap Lake, 2012

Variable	Detection Level (µg/g ww)	Variable	Detection Level (µg/g ww)
% Moisture	0.1	Molybdenum (Mo)	0.004
Aluminum (Al)	0.4	Nickel (Ni)	0.01
Antimony (Sb)	0.002	Phosphorus (P)	50
Arsenic (As)	0.004	Potassium (K)	200
Barium (Ba)	0.01	Rhenium (Re)	0.002
Beryllium (Be)	0.002	Rubidium (Rb)	0.01
Bismuth (Bi)	0.002	Selenium (Se)	0.02
Boron (B)	0.2	Silver (Ag)	0.001
Cadmium (Cd)	0.002	Sodium (Na)	200
Calcium (Ca)	5	Strontium (Sr)	0.01
Cesium (Cs)	0.001	Tellurium (Te)	0.004
Chromium (Cr)	0.01	Thallium (Tl)	0.0004
Cobalt (Co)	0.004	Thorium (Th)	0.002
Copper (Cu)	0.01	Tin (Sn)	0.004
Gallium (Ga)	0.004	Titanium (Ti)	0.01
Iron (Fe)	0.2	Uranium (U)	0.0004
Lead (Pb)	0.004	Vanadium (V)	0.004
Lithium (Li)	0.02	Yttrium (Y)	0.002
Magnesium (Mg)	10	Zinc (Zn)	0.1
Manganese (Mn)	0.004	Zirconium (Zr)	0.04
Mercury (Hg)	0.001	% Lipid Content	0.5

% = percent; µg/g ww = micrograms per gram wet weight.

9.2.2 Data Analyses

9.2.2.1 Data Handling

Prior to summarizing and performing statistical analyses on the fish tissue chemistry data, values below the limit of detection, or non-detects, were reviewed. Where data were below the laboratory detection limit, values were set to one-half the detection limit (USEPA 2000). If results for one parameter were all below the detection limit, no mean was calculated, and the result was reported as "not-detected".

Data Transformations and Statistical Outliers

Prior to statistical analyses, tests were performed to determine whether the data met parametric assumptions (i.e., were normally distributed and demonstrated equality of variance). All data were log_{10} transformed and subsequently screened as both raw (untransformed) data and log_{10}-transformed data. The goodness-of-fit of each dataset to a normal distribution was tested using a Kolmogorov-Smirnov test. The assumption of group variances being equal was tested using Levene's tests (Appendix 9A, Table 9A-1). A final data screening process was performed for covariate analyses (e.g., mercury and selenium) to confirm a significant regression relationship between the dependent (body weight or fish length) and independent (mercury or selenium concentration) variables (Appendix 9A, Table 9A-2).

The presence of statistical outliers within a dataset can greatly influence normality and equality of variance, and the resulting type of statistical test (i.e., parametric or non-parametric) that can be performed. Standardized residuals (SR) from linear regression analyses were used as a screening tool for identifying statistical outliers. Unexplained deviations from the regression line were quantified by calculating SR values. The SR considers leverage or influence of an observation on the regression, as well as the magnitude of the residual (Sokal and Rohlf 2012). In brief, if an observation shows high leverage, but has a low residual, this indicates consistency with the regression model and the observation is not unusual within the dataset. Large residuals with low leverage values are not considered unusual, as they do not have notable influence on the regression line. If, however, observations show both high leverage and high residuals (i.e., large SR values), then such observations may unduly affect the slope of the regression line (Sokal and Rohlf 2012). Observations that had $SR>|3|$ were checked and their validity confirmed. Once confirmed, these observations were considered statistical outliers and were removed from relevant statistical tests. Statistical testing was performed only on outlier-removed datasets following the SR outlier identification process described above. Statistical analyses were conducted using the software SYSTAT 13.1.05 (SYSTAT 2009).

9.2.2.2 Statistical Analyses

Descriptive statistics, including sample size, arithmetic mean, minimum, maximum, standard error (SE), and standard deviation (SD), were calculated by lake for each metal.

Only parameters that were elevated in fish tissue from Snap Lake relative to the reference areas were compared statistically. Analysis of Variance (ANOVA) and Analysis of Covariance (ANCOVA) was used to test for differences in the concentrations of metals measured in Lake Chub when data met parametric assumptions. For parameters that did not meet parametric test assumptions, the Kruskal-Wallis test was used to test for differences among sampling areas. Alpha (α) and beta (β) were set equal at 0.1 for all statistical analyses (Environment Canada 2012), resulting in a statistical power (i.e., $1-\beta$) of 90%.

ANOVA

If a significant difference was present among all lakes following ANOVA (i.e., probability (*P)*<0.1), a planned contrast was performed to test for differences between Snap Lake and the pooled reference lakes (i.e., Northeast Lake and Lake 13 combined). To test for differences between the reference lakes, paired contrasts comparing Lake 13 to Northeast Lake were also conducted. The magnitude of the difference between Snap Lake and the reference lakes for ANOVAs was calculated by expressing the difference as a percentage of the pooled mean of the two referenced lakes:

$$Magnitude\ Difference\ (\%) = \frac{(Exposure\ Mean - Pooled\ Reference\ Mean)}{Pooled\ Reference\ Mean} \times 100. \qquad \text{[Equation 9-1]}$$

The magnitude of the difference between reference lakes for ANOVAs was calculated as the relative percent difference:

$$Relative\ Percent\ Difference\ (\%) = \frac{I(Reference\ 1\ Mean - Reference\ 2\ Mean)I}{Pooled\ Reference\ Mean} \times 100. \qquad \text{[Equation 9-2]}$$

ANCOVA

Mercury and selenium biomagnify (i.e., accumulate via food up three or more trophic levels to a greater degree at each trophic level); therefore, these metals were standardized to fish size for statistical testing, and ANCOVA was used to test for differences among lakes. To determine the best covariate for fish size, a regression analysis for mercury and selenium concentration against each potential covariate (i.e., length or weight) was performed. The covariate with the strongest regression relationship (i.e., smallest *P*-value) was used as the covariate for the ANCOVA analysis.

Overall, neither length nor weight was a significant predictor of selenium concentrations in Lake Chub among lakes (i.e., the regression relationship was not significant); therefore, differences among lakes in selenium concentrations were tested by ANOVA (Appendix 9A, Table 9A-2). There was a significant linear relationship between mercury and fork length among lakes; therefore, ANCOVA was performed using length as the covariate (Appendix 9A, Table 9A-2).

An assumption of ANCOVA is that the slopes of the regression lines among treatment groups are equal; therefore, a test for homogeneity of slopes was conducted prior to performing ANCOVA analyses. If there was no significant interaction between sampling lakes and the covariate (i.e., assumption of homogeneity of slopes was satisfied), then an ANCOVA was performed and the adjusted means were calculated.

If a significant difference was determined in the ANCOVA analyses, a planned contrast was performed to test for differences between Snap Lake and the pooled reference lakes (i.e., Northeast Lake and Lake 13 combined). Paired contrasts comparing Lake 13 to Northeast Lake were also conducted. The magnitude of the differences among lakes for ANCOVAs was calculated with least squared means:

$$Magnitude\ Difference\ (\%) = \frac{(Exposure\ LSM - Reference\ LSM)}{Reference\ LSM} \times 100. \qquad \text{[Equation 9-3]}$$

The magnitude of the difference between reference lakes for ANCOVAs was calculated as the relative percent difference:

$$Relative\ Percent\ Difference\ (\%) = \frac{\mathrm{I}(Reference\ 1\ LSM - Reference\ 2\ LSM)}{Pooled\ Reference\ LSM} \times 100. \qquad \text{[Equation 9-4]}$$

9.2.2.3 Normal Range

Fish tissue chemistry was compared to the normal range (± 2SD of the pooled reference lake mean) to determine whether any changes seen in Snap Lake during the 2012 fish tissue chemistry survey were beyond the range of variability present in the reference lakes.

9.2.2.4 Guideline Comparison

Fish tissue metal concentrations were compared to available national guidelines for human health. The Canadian Food Inspection Agency (CFIA) and Health Canada guidelines state fish collected for commercial use may contain a maximum of 0.5 micrograms per gram wet weight (µg/g ww) mercury to be approved for human consumption (CFIA 2009). Arsenic and lead are subject to Health Canada guidelines concerning the sale of fishery products for human consumption: arsenic tissue concentrations must be below 3.5 microgram per gram wet weight (µg/g ww) in fish tissue for human consumption, while lead must be below 0.5 µg/g ww (CFIA 2009). While Lake Chub are not typically consumed by humans, and fish from Snap Lake are not sold as commercial fish, these guidelines were still used for comparison purposes.

9.3 QUALITY ASSURANCE AND QUALITY CONTROL

Quality assurance (QA) and quality control (QC) procedures were applied to field sampling, laboratory analyses, data entry, data analyses and report preparation tasks to produce technically sound and scientifically defensible results.

Field and laboratory equipment were calibrated throughout the field program as per specifications (i.e., daily or each use), and all samples were collected by experienced personnel. Samples were labelled, preserved, and shipped according to standard protocols. Specific work instructions outlining each field task in detail were provided to the field personnel by the task manager. Detailed field notes were recorded in waterproof field books and on pre-printed waterproof field data sheets in either pencil or indelible ink. Data sheets and sample labels were checked at the end of each field day for completeness and accuracy, and scanned into electronic copies at the completion of the field program. Chain-of-custody forms were used to track the shipment of samples.

Laboratory QA/QC included analysis of a series of sample blanks, spikes, and duplicates. ALS results indicated that relative percent differences for the laboratory duplicates of Lake Chub tissues were below 10% for all analytes. Additionally, internal laboratory blanks were below the limit of detection, and analyses of known concentration reference materials were all within the laboratories standard acceptable range of limits.

Data entry QA/QC involved checking a minimum of 10% of the data for completeness, data entry errors, transcription errors on field sheets, and invalid or impossible data values. If an error was found, data underwent a zero tolerance QC check, where every datum was checked.

Results of statistical analyses were independently reviewed by a senior biologist with appropriate technical qualifications. Tables containing data summaries and statistical results were reviewed, and values verified by a second, independent individual.

9.4 RESULTS

A total of 24 samples were analyzed for percent moisture content, metals, and lipid concentration (Appendix 9B). Summary statistics are provided in Appendix 9C, while summary figures (mean ± SD) for all parameters are provided in Appendix 9D.

9.4.1 Statistical Comparisons

Metals with concentrations below the detection limit for fish collected from Snap Lake were antimony, beryllium, gallium, rhenium, silver, tellurium, thorium, tin, yttrium, and zirconium. Therefore, statistical analysis could not be and did not need to be performed for these parameters. Some metal concentrations in Lake Chub collected from Snap Lake were lower than those measured in fish from the reference areas, specifically barium, calcium, lithium, magnesium, phosphorus, and selenium (Table 9-3). Since a decrease in metal concentrations in fish from Snap Lake is unlikely to be the result of exposure to treated effluent, only those metals showing an increase in tissue concentrations relative to the reference areas are presented in Tables 9-4 and 9-5.

Table 9-3 Concentration (Mean ± SD, µg/g ww) of Metals in Lake Chub From Lake 13, Northeast Lake, and Snap Lake, 2012

Variable	DL	Lake 13	NEL	Snap Lake
Number of Samples	-	8	8	8
% Moisture	0.1	74.9 ± 2.23	76 ± 1.3	75.3 ± 2.3
Aluminum (Al)	0.4	1.0 ± 0.5 (1<DL)[(a)]	1.4 ± 1.1	1.3 ± 1.0
Antimony (Sb)	0.002	0.001 ± 0.001 (7<DL)	0.001 ± 0.001 (7<DL)	0.001 ± 0.000 (7<DL)
Arsenic (As)	0.004	0.019 ± 0.007	0.025 ± 0.006	0.022 ± 0.010
Barium (Ba)	0.01	3.52 ± 1.48	6.16 ± 2.17	2.02 ± 0.90
Beryllium (Be)	0.002	<DL[(b)]	<DL	<DL
Bismuth (Bi)	0.002	0.004 ± 0.004 (3<DL)	0.002 ± 0.001 (6<DL)	0.012 ± 0.015 (3<DL)
Boron (B)	0.2	0.2 ± 0.2 (6<DL)	0.05 ± 0.5 (5<DL)	0.2 ± 0.2 (6<DL)
Cadmium (Cd)	0.002	0.010 ± 0.004	0.017 ± 0.005	0.016 ± 0.011
Calcium (Ca)	5	15211 ± 6579	12685 ± 3866	11531 ± 4486
Cesium (Cs)	0.001	0.021 ± 0.007	0.034 ± 0.008	0.033 ± 0.012
Chromium (Cr)	0.01	0.02 ± 0.01 (1<DL)	0.02 ± 0.01	0.02 ± 0.02 (1<DL)
Cobalt (Co)	0.004	0.006 ± 0.003 (2<DL)	0.009 ± 0.004	0.012 ± 0.008
Copper (Cu)	0.01	0.41 ± 0.11	0.43 ± 0.10	0.48 ± 0.13
Gallium (Ga)	0.004	<DL	<DL	<DL
Iron (Fe)	0.2	7.8 ± 2.8	10.2 ± 3.4	9.3 ± 1.9
Lead (Pb)	0.004	0.002 ± 0.001 (7<DL)	0.003 ± 0.002 (6<DL)	0.003 ± 0.001 (6<DL)
Lithium (Li)	0.02	0.04 ± 0.02 (1<DL)	0.04 ± 0.01	0.01 ± 0.01 (6<DL)
Magnesium (Mg)	10	486 ± 119	425 ± 69	402 ± 75
Manganese (Mn)	0.004	5.313 ± 2.611	5.904 ± 1.914	6.863 ± 3.357
Mercury (Hg)	0.001	0.059 ± 0.021	0.094 ± 0.039	0.102 ± 0.049
Molybdenum (Mo)	0.004	0.010 ± 0.005	0.011 ± 0.005	0.013 ± 0.006
Nickel (Ni)	0.01	0.03 ± 0.02	0.03 ± 0.02	0.05 ± 0.02
Phosphorus (P)	50	9029 ± 3090	7741 ± 1790	7139 ± 2159
Potassium (K)	200	3218 ± 414	3281 ± 201	3315 ± 113
Rhenium (Re)	0.002	<DL	<DL	<DL
Rubidium (Rb)	0.01	4.72 ± 1.33	5.48 ± 1.06	4.59 ± 1.24
Selenium (Se)	0.02	0.45 ± 0.17	0.41 ± 0.05	0.38 ± 0.08
Silver (Ag)	0.001	<DL	<DL	<DL
Sodium (Na)	200	816 ± 196	1023 ± 91	971 ± 141
Strontium (Sr)	0.01	33.26 ± 12.55	39.63 ± 10.61	49.61 ± 15.79
Tellurium (Te)	0.004	<DL	<DL	<DL
Thallium (Tl)	0.0004	0.0018 ± 0.005	0.0024 ± 0.0007	0.0044 ± 0.0017
Thorium (Th)	0.002	0.001 ± 0.001 (7<DL)	<DL	<DL
Tin (Sn)	0.004	<DL	0.003 ± 0.002 (5<DL)	<DL
Titanium (Ti)	0.01	0.05 ± 0.03	0.10 ± 0.14	0.11 ± 0.11
Uranium (U)	0.0004	0.0058 ± 0.0037	0.0078 ± 0.0034	0.0123 ± 0.0116
Vanadium (V)	0.004	0.004 ± 0.002 (3<DL)	0.005 ± 0.005 (5<DL)	0.007 ± 0.004 (2<DL)
Yttrium (Y)	0.002	<DL	0.001 ± 0.001 (7<DL)	<DL
Zinc (Zn)	0.1	48.2 ± 12.2	51.6 ± 10.3	53.8 ± 16.3
Zirconium (Zr)	0.04	<DL	<DL	<DL
Lipid Content	0.5	2.5 ± 1.2	1.8 ± 0.6	2.4 ± 1.0

(a) DLs were replaced with 0.5×DL (see text; number of samples <DL shown in brackets).

(b) <DL = All samples had concentrations below the DL.

"-" = not applicable; DL = detection limit; SD = standard deviation; µg/g = microgram per gram; % = percent; <= less than; ± = plus minus; NEL = Northeast Lake.

Table 9-4 Statistical Comparisons[(a)] of Lake Chub Tissue Metal Concentrations among Sampling Areas

Variable	Statistical Test[(b)]	Differences among All Sampling Areas	Reference vs. Exposure Comparisons		Reference vs. Reference Comparisons	
			Snap vs. Lake 13 and NEL		Lake 13 vs. NEL	
		P[(c)]	*P*[(c)]	%[(d)]	*P*[(c)]	%[(e)]
Aluminum (Al)	ANOVA[log]	ns	nd	-1.33	nd	34.77
Arsenic (As)	ANOVA	ns	nd	-0.17	nd	24.24
Bismuth(Bi)	KW	*	ns	311.23	ns	88.55
Boron (B)	KW	ns	nd	-36.70	nd	88.93
Cadmium (Cd)	ANOVA[log]	*	ns	19.70	*	50.49
Calcium (Ca)	ANOVA	ns	nd	-17.33	nd	18.11
Cesium (Cs)	ANOVA[log]	**	ns	17.60	**	48.47
Chromium (Cr)	ANOVA[log]	ns	nd	24.09	nd	8.76
Cobalt (Co)	ANOVA[log]	ns	nd	56.12	nd	43.39
Copper (Cu)	ANOVA	ns	nd	15.04	nd	4.85
Iron (Fe)	ANOVA	ns	nd	3.81	nd	26.21
Lead (Pb)	KW	ns	nd	7.25	nd	25.12
Manganese (Mn)	ANOVA[log]	ns	nd	22.37	nd	10.54
Mercury (Hg)	ANCOVA[log]	*	ns	53.73	ns	47.76
Molybdenum (Mo)	ANOVA[log]	ns	nd	27.91	nd	10.97
Nickel (Ni)	ANOVA[log]	ns	nd	52.54	nd	0.85
Potassium (K)	ANOVA	ns	nd	2.68	nd	1.96
Rubidium (Rb)	KW	ns	nd	-10.00	nd	14.81
Sodium (Na)	ANOVA	*	ns	5.64	*	22.43
Strontium (Sr)	ANOVA[log]	*	*	36.13	ns	17.46
Thallium (Tl)	ANOVA[log]	***	***	109.62	ns	26.26
Titanium (Ti)	ANOVA[log]	ns	nd	42.71	nd	63.86
Uranium (U)	ANOVA[log]	ns	nd	80.39	nd	29.47
Vanadium (V)	KW	ns	nd	59.44	nd	26.11
Zinc (Zn)	ANOVA[log]	ns	nd	7.79	nd	6.82

(a) Comparisons were made only for those parameters that demonstrated increases in Snap Lake relative to the reference lakes based on graphical analyses.

(b) ANOVA = Analysis of Variance (log-transformed data indicated by superscript); ANCOVA = Analysis of Covariance (log-transformed data indicated by superscript); KW = Kruskal Wallis test.

(c) Overall probability of Type 1 Error: * = $P<0.1$, ** = $P<0.01$, *** = $P<0.001$, ns = non-significant.

(d) Percent difference between group means.

(e) Relative percent difference between group means.

nd = not determined as overall *P* not significant; *P* = probability; NEL = Northeast Lake; % = percent.

Table 9-5 Normal Range Observed for Lake Chub Tissue Chemistry Parameters in Reference Areas and Mean of Observed Data in Snap Lake

Variable	Reference Areas Lake 13 + NEL	Snap Lake
	Normal Range (µg/g ww; Mean ± 2SD)	Mean (µg/g ww)
Aluminum (Al)	0 - 2.3	1.3
Arsenic (As)	0.009 - 0.035	0.021
Bismuth (Bi)	0 - 0.009	0.012
Boron (B)	0 - 1.157	0.204
Cadmium (Cd)	0.0 - 0.0	0.0
Calcium (Ca)	3201 – 24695	11531
Cesium (Cs)	0.008 - 0.048	0.033
Chromium (Cr)	0 - 0.04	0.02
Cobalt (Co)	0 - 0.015	0.012
Copper (Cu)	0.21 - 0.62	0.48
Iron (Fe)	2.5 - 15.5	9.3
Lead (Pb)	0 - 0.005	0.003
Manganese (Mn)	1.143 - 10.073	6.863
Mercury (Hg)	0.006 - 0.146	0.102
Molybdenum (Mo)	0 - 0.021	0.013
Nickel (Ni)	0 - 0.06	0.05
Potassium (K)	2617 - 3882	3315
Rubidium (Rb)	2.64 - 7.55	4.59
Sodium (Na)	556 - 1283	971
Strontium (Sr)	13.81 - 60.60	49.61
Thallium (Tl)	0.0007 - 0.0030	0.0040
Titanium (Ti)	0 - 0.27	0.11
Uranium (U)	0 - 0.0140	0.0120
Vanadium (V)	0 - 0.012	0.007
Zinc (Zn)	27.83 - 71.95	53.78

SD = standard deviation; µg/g ww= microgram per gram wet weight; NEL = Northeast Lake.

Most metals were found to be statistically similar among all lakes. The concentrations of seven metals were significantly different among all lakes: bismuth; cadmium; cesium; mercury; sodium; strontium; and, thallium (Table 9-4). Of these metals, only strontium and thallium concentrations were significantly higher in Lake Chub from Snap Lake relative to the reference lakes. Cadmium, cesium, and sodium were significantly higher in Northeast Lake compared to Lake 13. Bismuth and mercury were marginally significantly different among all lakes and, as a result, a statistical difference was not observed with the individual lake contrasts.

9.4.2 Normal Range

Thallium and bismuth concentrations were greater than the normal range (Table 9-5). While the mean bismuth concentration in Snap Lake was greater than the normal range, a statistical difference between Snap Lake and the reference lakes was not observed (Tables 9-4 and 9-5). This is likely due to the large number of non-detect bismuth values from each lake, as three of the Snap Lake samples and half of the reference lake samples had bismuth concentrations below the detection limit. This is also reflective of why there is a large, but non-significant, magnitude percent difference between Snap Lake and the reference lakes (Table 9-4).

9.4.3 Guideline Comparison

Mean mercury concentration in Lake Chub captured in all lakes was below the CFIA (2009) guideline of 0.5 µg/g ww. Lake Chub captured in all lakes had concentrations below the CFIA (2009) arsenic guideline of 3.5 µg/g ww and lead guideline of 0.5 µg/g ww for commercial sale of fish.

9.5 SUMMARY

Concentrations of most metals, with the exception of strontium and thallium, were statistically similar between Snap Lake and the reference lakes, and were within the normal range and/or below their respective DL in Lake Chub.

Strontium concentrations in fish carcasses in Snap Lake ranged from 26.7 to 77.1 µg/g ww. This range is overlapping, though higher than, the range of concentrations seen in other small-bodied fish (i.e., Slimy Sculpin [*Cottas cognatus*]) exposed to diamond mining activities (19.0 to 40.8 µg/g ww; Golder 2011). Adverse effects to fish at tissue concentrations presently reported in Lake Chub were not observed in 2012 (See Section 7.11).

Thallium concentrations in Snap Lake Lake Chub fish carcasses ranged from 0.0017 to 0.0065 µg/g ww. These concentrations are considerably lower than those seen in other small-bodied fish (i.e., Slimy Sculpin) from reference locations in the Northwest Territories (0.0073 to 0.0147 µg/g ww; Golder 2011). Thallium concentrations are very low to non-detectable in the treated effluent at Snap Lake (Appendix 3D, Figure 3D-45) and in Snap Lake main basin and Snap Lake sediment (Section 3.5; Section 4.5; Appendix 3F, Figure 3F-47). Increased concentrations of thallium relative to reference have not been previously observed in either Lake Trout or Round Whitefish (De Beers 2012b) tissue samples; therefore, the source of higher concentrations of thallium in Lake Chub captured in Snap Lake is uncertain. Monitoring of thallium in fish tissue will continue in 2013 and 2015.

For many parameters, the magnitude of difference observed between Snap Lake and the reference lakes was similar to the magnitude of difference between the two reference lakes. The differences in metal concentrations between reference lakes are likely the result of natural variability among populations, and cannot be attributed to the Mine. The addition of the second reference lake to the 2012 AEMP highlights the natural variability among populations, and helps to distinguish natural variability versus Mine-related effects.

9.6 CONCLUSION

9.6.1 Key Question: Are Tissue Metal Concentrations in Fish From Snap Lake Increasing Relative to Reference Lakes?

Both strontium and thallium were significantly different between Snap Lake and the reference lakes. The difference in strontium was not greater than normal range. The difference in thallium concentration between Snap Lake and the reference lakes was greater than normal range. Despite the elevated concentrations of these two metals in Lake Chub from Snap Lake, there was no evidence of negative effects on fish health (see Section 7.11).

9.7 RECOMMENDATIONS

The following recommendations are provided for consideration in the future Snap Lake fish tissue chemistry AEMP program:

Continuation of the small-bodied tissue survey - This study represents the first small-bodied fish tissue survey in Snap Lake. This survey should continue every three years to provide an early indicator of potential changes in large-bodied fish, and to support potential effects observed during the small-bodied fish health survey.

Review thallium in 2013 large-bodied fish survey - As the source of increased thallium concentration in Lake Chub collected from Snap Lake is uncertain, trends over time in thallium concentrations for Lake Trout and Round Whitefish should be reviewed during the 2013 large-bodied fish tissue chemisty survey.

9.8 REFERENCES

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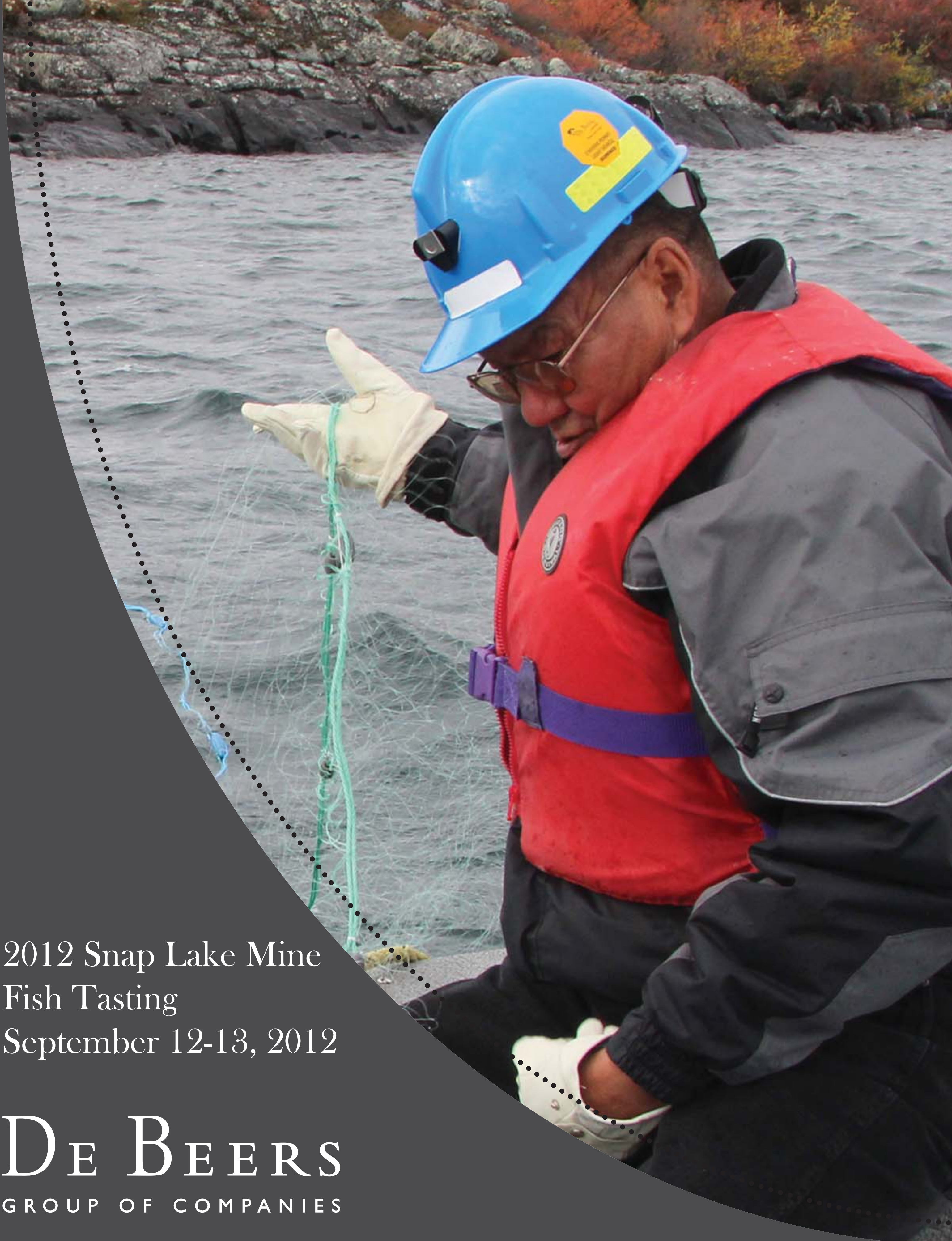
10 FISH TASTING



2012 Snap Lake Mine Fish Tasting
September 12-13, 2012

DE BEERS
GROUP OF COMPANIES

Elders gather round to inspect a fish being gutted during the 2012 fish tasting at the Snap Lake Mine.

Thank You to All Participants

De Beers wants to take this opportunity to thank the elders and everyone who took part in the 2012 Fish Tasting:

Elders & Interpreters

Tlicho Government
Nick Football - Elder

Lutsel K'e Dene First Nation
Joe Catholique - Elder
John Catholique - Elder
Archie Catholique – Interpreter

North Slave Métis Alliance
Hugh McSwain - Elder
Wayne Langenhan – Elder

Yellowknives Dene First Nation
Mike Francis - Elder
Philip Liske – Elder
Lena Drygeese - Interpreter

Snap Lake Environmental Monitoring Agency
Dave White – Executive Director
Zhong Lui – Environmental Analyst

De Beers Canada
Maxwell Morapeli - Mine General Manager
Herman Henning - Mining Manager

Tom Bradbury – Permitting Coordinator
Sabet Biscaye – Superintendent, Community Relations
Terry Kruger – Sr. Communications Specialist
Erin Rowlands – Environmental Coordinator
Andrea Hrynkiw - Environmental Technician

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Annual Snap Lake Fish Tasting Shares Traditional Knowledge

Traditional knowledge is important to De Beers.

That's why during the Environmental Assessment of the Snap Lake Mine, we committed to holding annual fish tasting events at the mine. Each year since 2005, we have brought elders from communities close by to catch fish from the lake. They use their lifetime of experience and knowledge to examine the fish to see if they look normal and healthy.

Fillets of the fish are boiled in water, then eaten without addition of salt, pepper, oil or butter, to allow elders to fully taste the flavour of the fish. Elders' comments are documented in this publication.

During the 2012 fish tasting, Sept. 12-13, elder came from Lutsel K'e Dene First Nation, Yellowknives Dene First Nation, Tlicho Government and the North Slave Métis Alliance.

If you have any questions or comments, please contact me.

A. Hood

Alexandra Hood
Superintendent, Environment & Permitting

Elder: Philip Liske (YKDFN)	
Fish Health Observation: Fish health is above average	Very Good
Comment: "The skin is not flabby and is nice and smooth. It looks healthy from the outside." The eggs are a little dark. The guts are somewhat dark.	
Texture Observation: Fish is above average quality	Good
Comment: "It's nice and firm ... healthy. Not greasy."	
Fish Taste Observation: Fish taste appears to be average	Good
Comment: "The fish was a little flat tasting."	

Elder: Mike Francis (YKDFN)	
Fish Health Observation: appears to be of average health	Good
Comment: "The fish looks good. The health is good."	
Texture Observation: Texture is average firmness	Good
Comment: N/A	
Fish Taste Observation: Fish taste appears to be average	Good
Comment: "It tastes good."	

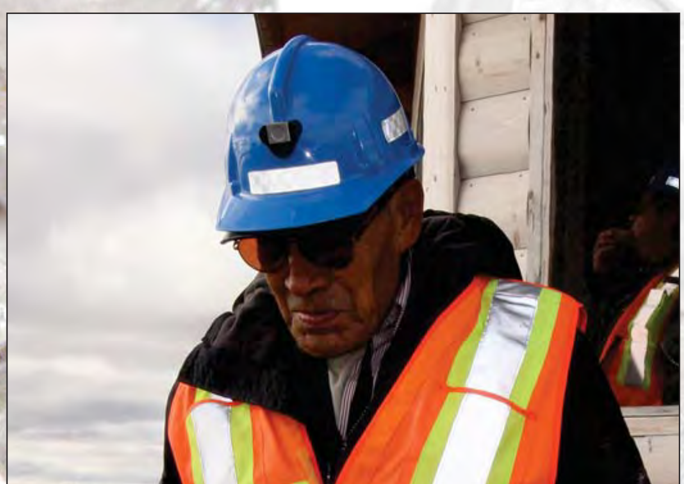
Elder: Nick Football (Tlicho)	
Fish Health Observation: Appears to be above health	Good
Comment: "It looks good. It appears dark, but inland lakes have fish with darker colour inside."	
Texture Observation: Fish is above average quality	Very Good
Comment: "It has really good texture. Nothing bad to say."	
Fish Taste Observation: Fish taste appears to be average	Good
Comment: N/A	

Elder: Archie Catholique (LKDFN interpreter)	
Fish Health Observation: Appears to be of average health	Good
Comment: "The skin colour was good. The eggs were dark, but the liver and head were good."	
Texture Observation: Texture is average firmness	Good
Comment: "The texture was a little soft, possibly because the water was shallow."	
Fish Taste Observation: Fish taste appears to be average	Good
Comment: "The taste was good. No complaints."	

Elder: Wayne Langenhan (NSMA)	
Fish Health Observation: Appears to be of average health	Good
Comment: N/A	
Texture Observation: Texture is average firmness	Good
Comment: "It's firm enough."	
Fish Taste Observation: Fish taste appears to be average	Good
Comment: "The taste was good. It's different than the fish from Diavik."	

Elder: Hugh McSwain (NSMA)	
Fish Health Observation: Appears to be of average health	Good
Comment: "The taste was good."	
Texture Observation: Texture is average firmness	Good
Comment: "The texture was good."	
Fish Taste Observation: Fish taste appears to be average	Good
Comment: "Overall, the fish looks good to me."	

Elder: John Catholique (LKDFN)	
Fish Health Observation: Appears to be of average health	Good
Comment: "It might be dark because it's inland fish. Looks healthy"	
Texture Observation: Texture is average firmness	Good
Comment: "Good texture."	
Fish Taste Observation: Fish taste appears to be average	Good
Comment: "The taste was good, almost like Great Slave Lake fish."	



Elder: Joe Catholique (LKDFN)	
Fish Health Observation: Appears to be of average health	Good
Comment: "Everything looks good. The fish looked healthy. The liver, heart and skin were good. We are further north so the fish is a different colour."	
Texture Observation: Texture is average firmness	Good
Comment: N/A	
Fish Taste Observation: Fish taste appears to be average	Good
Comment: "Inland lake fish taste different, but the fish was better."	

Elder: Lena Drygeese (YKDFN interpreter)	
Fish Health Observation: Appears to be of average health	Good
Comment: "The flesh is the proper colour for inland fish."	
Texture Observation: Texture is average firmness	Good
Comment: N/A	
Fish Taste Observation: Fish taste appears to be average	Good
Comment: "There was no fat, which is good. It tasted good because of the way it was cooked."	

Conclusions of 2012 Snap Lake Mine Fish Tasting

The annual fish tasting event at the Snap Lake Mine was held Sept. 12-13, with seven elders and two interpreters taking part. They were joined by observers from the Snap Lake Environmental Monitoring Agency. Mine General Manager Maxwell Morapeli and Mining Manager Herman Henning also joined the group for part of the day.

Each elder was able to fully participate in the tasting, even though only two fish were caught. All elders and interpreters rated the fish Good or Very Good in all three observational categories.

Compiled Results - 2012 Snap Lake Mine Fish Tasting			
Fish Health Observation	Very Good - 1	Good - 8	Not Good - 0
Fish Texture Observation	Very Good - 1	Good - 8	Not Good - 0
Fish Taste Observation	Very Good - 0	Good - 9	Not Good - 0

Photo Gallery

Archie Catholique, right, tends the fish on the fire as Snap Lake Mine General Manager Maxwell Morapeli, centre, and Philip Liske chat.

(Above) Hugh McSwain hands a lighter to Wayne Langenhan as he lights the cook fire. (Below) A pair of fish heads roast over the coals.

An elder uses skill learned over many years to cut apart a fish head during the Snap Lake Mine fish tasting in September 2012.

Flames start to consume firewood during preparation for the annual fish tasting at the Snap Lake Mine.

11 TRADITIONAL KNOWLEDGE

The Water Licence (MV2011L2-0004 [Part G, Schedule 6, Item 2 (d)] requires that the Aquatic Effects Monitoring Program (AEMP) include:

A summary of how Traditional Knowledge has been collected and incorporated into the AEMP, as well as a summary of how Traditional Knowledge will be incorporated into further studies relating to the AEMP.

The primary objectives of the Traditional Knowledge (TK) component of AEMP for the Snap Lake Mine are to:

- meet Water Licence requirements;
- include TK with scientific knowledge in the design and implementation of the AEMP; and,
- recommend changes to the AEMP for future years.

The revised scope of the TK component is not yet finalized. The fish tasting program (Section 10) is a TK component and has been incorporated since 2004. De Beers would like to expand the TK component with additional community involvement in the 2013 AEMP Design Plan and the field programs. A preliminary meeting with communities was held on September 19, 2012. Community visits are scheduled for May/June 2013. Analyses and interpretation of the TK component will focus on answering key questions; these key questions will be developed in consultation with community members in mid-2013 and reported in the 2013 AEMP Annual Report.

12 SPECIAL STUDIES

12.1 LITTORAL ZONE SPECIAL STUDY

12.1.1 Introduction

A preliminary assessment of the littoral zone of Snap Lake was conducted in this special study in 2012, focusing mainly on determining the best means of sampling this zone. This Special Study will continue for two more years (i.e., further work will be conducted in 2013 and 2014). Full data evaluation will occur after the full three years of this Special Study (i.e., as part of the 2014 Annual Aquatic Effects Monitoring Program [AEMP] Report).

12.1.1.1 Background

Importance of the Littoral Zone

The littoral zone is one of the most diverse and complex areas of lake ecosystems (Turner 1993). It is the near-shore region and is the link between the catchment area of the lake and its open-water (Wetzel 2001). Unlike the open-water, which is relatively homogenous, the littoral zone can be a heterogenous assemblage of surfaces. This diverse area is capable of supporting a wide range of independent and sometimes interconnected communities, which include plants, attached algae, bacteria, protozoans, sessile invertebrates, motile grazers and scrapers, seasonally important egg-laying fish, and other small transient fish species (Moss 2005).

As the link between the catchment area and the open-water, the littoral zone may be affected by, and in turn influence, what is occurring in the open-water. The littoral zone acts as an interceptor or a sink for nutrients, and as a source of new nutrients (Moss 2005). Since it can act as both a sink and source, it can increase the residence time of nutrients in the lake (Riber et al. 1983). In contrast to the open-water, which typically requires a sustained input of new nutrients for algal growth (Wetzel 2001), the littoral zone has the capacity to recycle and retain an internal nutrient load (Riber et al.1983; Turner et al. 1994).

Littoral zones can be important to lake health and productivity. The size of the littoral zone in relation to the size of the pelagic region varies greatly among lakes and depends on the geomorphology of the lake. In relatively small lakes, such as Snap Lake, littoral flora can contribute substantially to lake productivity and may even dominate and regulate the metabolism of the entire lake ecosystem (Wetzel 2001). The physiological and ecological characteristics of this zone provide habitat for photosynthetic and heterotrophic microflora, as well as zooplankton and larger invertebrates. These communities, in turn, are part of the food chain leading up to fish and are important to the health and sustainability of the fish community in the lake (Hille 2008). In addition, the littoral flora synthesize large quantities of organic matter, most of which accumulates in the sediments (Wetzel 2001).

As lakes receive nutrients, there is a tendency for phytoplankton biomass and abundance to increase within the limitations of temperature and available light, which can result in eutrophication. Periphyton in the littoral zone is a complex mixture of algae, bacteria, and detritus attached to submerged surfaces in most aquatic ecosystems, called the epilithon (Wetzel 2001). Littoral periphyton productivity can, in small lakes, result in increased eutrophication of the lake system as a whole.

Littoral Nutrients

The attached algal component of rock-associated periphyton, which is called epilithic algae (Wetzel 2001), obtains nutrients mostly by diffusion from the overlying water (Kahlert and Petterson 2002). Therefore, nutrient concentrations in the surrounding water influence the epilithic algae to a greater extent than plant-associated or sediment-associated periphyton.

As in the case of phytoplankton, the primary nutrients necessary for the development of epilithic algae are phosphorus (P), nitrogen (N), carbon (C), and, for diatoms, silica (Si). In lakes, P is often the limiting nutrient for phytoplankton (Schindler 1974, 1978); however, the role of P-limitation is less clear. Turner et al. (1994) showed that, in oligotrophic lakes of low alkalinity in the Experimental Lakes Area (ELA) in Ontario, rates of epilithic algal productivity can be limited by low concentrations of dissolved inorganic carbon (DIC) in the overlying water, rather than by P. The supply of DIC for photosynthesis is restricted by the boundary layer, a layer of inactive water above the epilithic algae (Kahlert and Petterson 2002). The thicker the boundary layer, the slower the exchange, although this layer may also allow trapping and recycling of P (Riber and Wetzel 1987). As a result, even though P is ultimately the limiting nutrient for epilithic algae, there may be a shift to C-limitation in low DIC lakes because P accumulates and is recycled. Conversely, when DIC concentrations are elevated in the overlying water, the epilithic algae can be P-limited (Turner et al. 1994). When DIC is low in the overlying water, increased P-loading will favour energy flow in the pelagic zone rather than the littoral zone of a lake (Turner et al. 1994).

Epilithic algae compete with phytoplankton for light and nutrients (Vadeboncoeur et al. 2002). In high DIC lakes, where P remains limiting, an increase in P-loading can initially increase epilithic algal biomass and productivity (Fairchild and Lowe 1984; Cattaneo 1987). Water motion alters the physicochemical environment of the boundary layer, and can result in depletion of P and other nutrients in the littoral zone (Stevenson et al. 1982; Turner et al. 1994). In contrast, thick, dense, and active epilithic algae in standing waters will typically have relatively high nutrient concentrations (Sand-Jensen 1983).

The approximate molar ratios that the nutrients in the epilithic algae and associated bacteria and detritus can be found in are reflected in the Redfield Ratio (molar ratio of 106C:16N:1P; Wetzel 2001). The cellular C:N ratio can be used to indicate both N limitation and nutrient limitation in general (Healey and Hendzel 1980). The C:P ratio may be a good index of P-limitation (Healey and Hendzel 1980) and of the food quality of the epilithic algae to littoral grazers (Elser et al. 2000), while the cellular N:P ratio may distinguish between N- and P-limitation (Hillebrand and

Sommer 1999). Cellular nutrient concentrations of natural epilithic algal communities can reveal the type and extent of nutrient limitation and requirements. However, cellular nutrients, as well as other available and unavailable nutrients are available to epilithic algae via the complex mixture of bacteria and detritus that they are part of within the periphyton, rendering interpretation of such ratios arguably more complex in the littoral than in the pelagic zone of lakes.

Littoral Invertebrates

Littoral invertebrates are small aquatic animals that lack backbones, and live in the near-shore area of lakes, typically insect larvae, crustaceans, worms, leeches, snails, and clams. Littoral invertebrates provide a food source to seasonally important egg-laying fish and other small transient fish species (Moss 2005). They can influence epilithic algal biomass and community composition (Lamberti and Moore 1984), and can act as a link between primary producers and fish (Wetzel 2001).

Littoral invertebrates form diverse communities, and can consist of thousands of individuals per square metre. They live on the bottom of the sediments in the near-shore region (e.g., among, under, or on rocks; burrowing into or on the surface of sediments, and associated with aquatic plants). Snails with a heavy shell and strong muscular foot are common if calcium concentrations in the water are high, which is the case in Snap Lake.

Littoral invertebrates can affect epilithic algal community composition and biomass through grazing pressure. They can be useful for monitoring the environmental status of shallow lakes (Rosenberg and Resh 1993).

12.1.1.2 Littoral Zone Special Study in Snap Lake

A periphyton monitoring program was completed in 2004 as a Special Study to fulfill the requirements, of the Mackenzie Valley Land and Water Board (MVLWB) Class A Water Licence (Water Licence No. MV2001 L2-0002: MVLWB 2012). The Water Licence required De Beers to monitor periphyton biomass and community composition in Snap Lake to determine whether this community was being affected by the Snap Lake Mine (the Mine).

The 2004 Special Study was designed to assess the feasibility of periphyton sampling in Snap Lake, focusing on epilithic algae, and to gather baseline information to which future monitoring data could be compared. The results of the 2004 study indicated that sampling was difficult in Snap Lake due to logistical concerns (De Beers 2005). It was not recommended for future inclusion in AEMP monitoring and was subsequently removed as a Water Licence requirement.

The 2012 Littoral Zone Special Study was initiated in Snap Lake following recent AEMP findings that an apparent enrichment effect in the plankton and deep water benthic invertebrate communities was observed, without changes in total phosphorus (TP) concentrations in the water quality. This could indicate phosphorous being intercepted and retained by the littoral zone, with consequent increased productivity in that zone reflected throughout Snap Lake. In addition, high total organic carbon concentrations in the sediments (i.e., close to 20 percent [%]) and low dissolved oxygen concentrations may be an indication of littoral zone material affecting other areas of Snap Lake.

12.1.1.3 Objectives

The overall goal of the Littoral Zone Special Study was to explore the feasibility of conducting littoral zone monitoring as part of the AEMP in future. Specific objectives were to:

- determine the most appropriate and cost effective method for potential routine sampling during the Snap Lake Mine AEMP;
- determine the importance of the littoral zone to overall productivity in Snap Lake and Northeast Lake, which was used as a reference area;
- evaluate whether any changes have occurred in the epilithic algal community since the 2004 baseline study;
- investigate differences in the littoral invertebrate and epilithic algal communities between Northeast Lake and Snap Lake;
- determine whether within-station and within-lake variability are low enough to assess possible Mine-related effects; and,
- determine whether littoral data can provide useful, necessary new information that cannot be obtained from existing AEMP components.

The Littoral Zone Special Study was based on four key questions. These questions were modified slightly from the 2013 AEMP Design Plan (De Beers 2012a) after consultation with an external expert, Dr. Michael Turner (a retired Fisheries and Oceans Canada research scientist with extensive national and international expertise in littoral zone scientific studies):

1. Can littoral monitoring be conducted in Snap Lake and Northeast Lake, and does the inherent variability in the littoral zone allow the detection of Mine-related changes?
2. What are the current ratios of particulate C:N, C:P, N:P, and C:chlorophyll *a*, and what is the current percent algal carbon in the littoral zones of the main basin of Snap Lake, northwest arm of Snap Lake, and Northeast Lake? How do these values compare to baseline and what do these values indicate about Mine-related changes in nutrient status and food quality for invertebrates and fish?
3. What is the current status, in terms of relative abundance and relative biomass, of the epilithic algal communities in the main basin of Snap Lake, northwest arm of Snap Lake, and Northeast Lake? Do these results provide any evidence of a Mine-related effect?
4. What is the current invertebrate composition in the littoral zones of the main basin of Snap Lake, northwest arm of Snap Lake, and Northeast Lake? Do these results provide any evidence of a Mine-related effect?

12.1.2 Methods

12.1.2.1 Sampling Locations and Timing

The Littoral Zone Special Study was designed on the basis of the 2004 baseline Periphyton Special Study in Snap Lake. A subset of the 2004 stations was re-sampled in Snap Lake and new stations were selected in Northeast Lake to collect reference data. Five stations in the main basin of Snap Lake, two stations in the northwest arm of Snap Lake, and three stations in Northeast Lake were sampled in 2012 (Figure 12.1-1; Table 12.1-1). Each epilithic algae and littoral invertebrate sampling station was located on cobble/boulder substratum within the euphotic zone, at a sampling depth of 2 metres (m) (i.e., below the wave-washed zone).

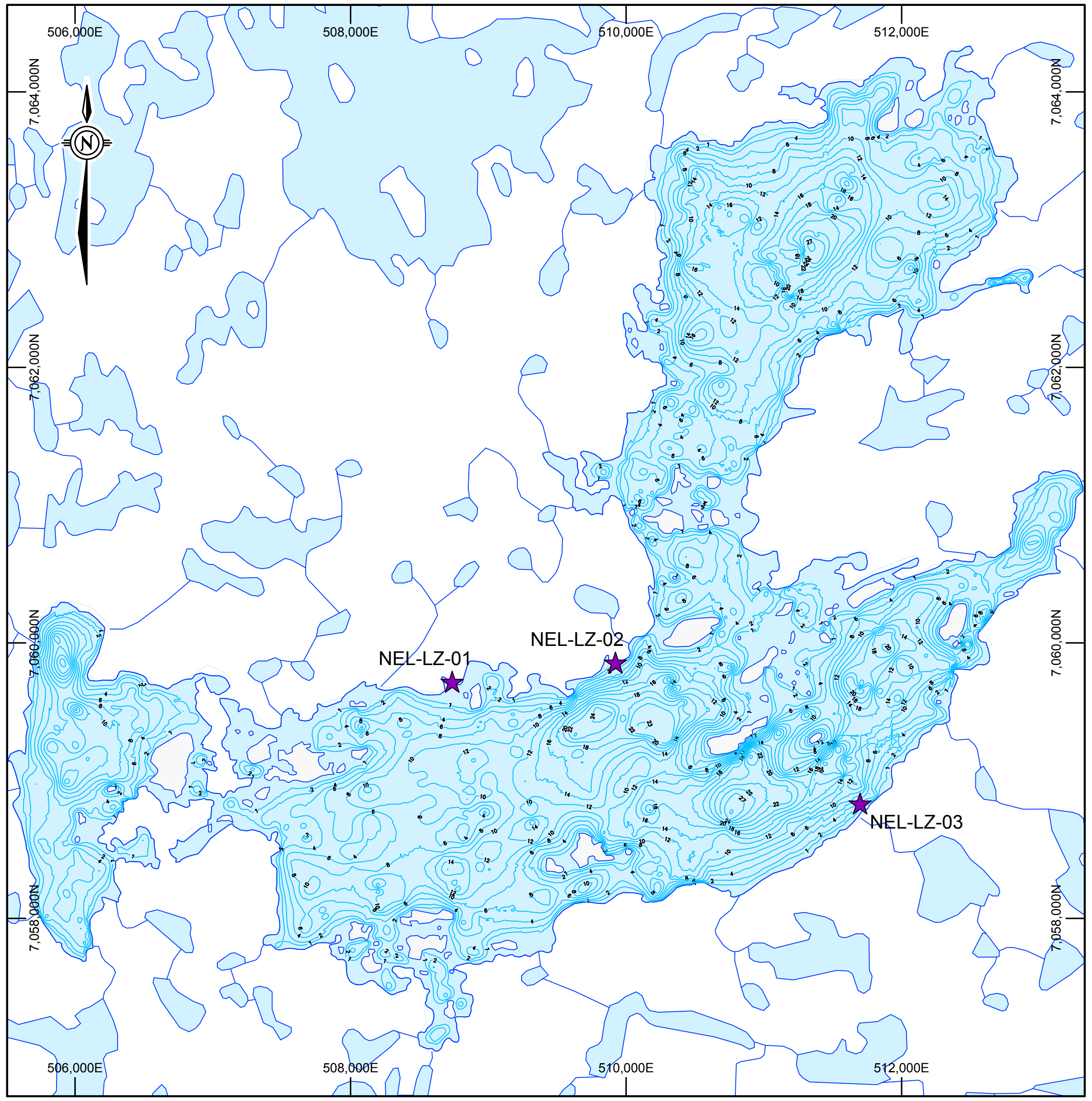
Samples were collected from August 12 to 15, 2012. A mid-August sampling period was selected to replicate the timing of the 2004 Periphyton Special Study and to allow sampling during the period of maximum productivity, which is typically August in the sub-Arctic region.

Table 12.1-1 Littoral Zone Sampling Stations in Snap Lake and Northeast Lake

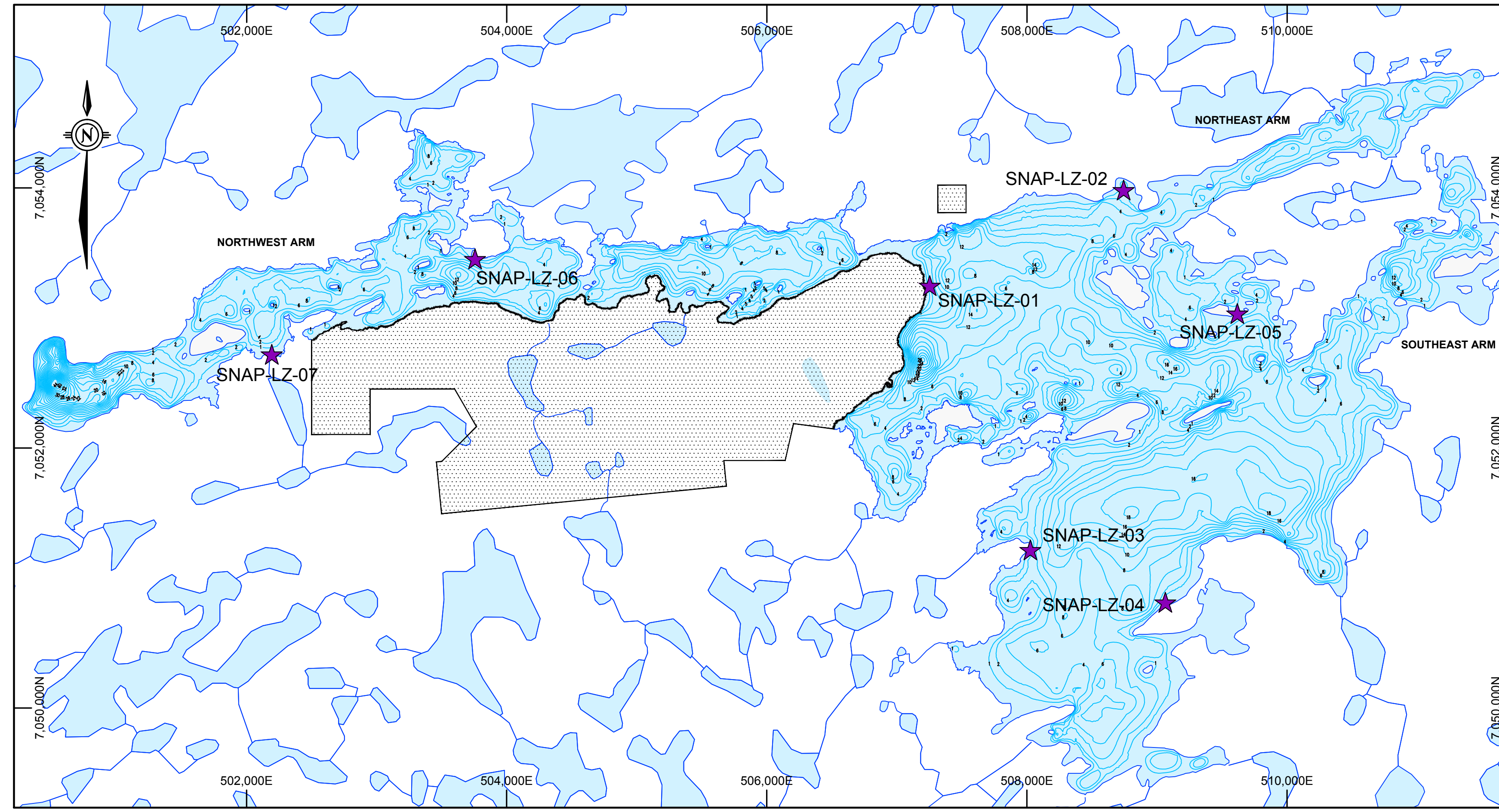
Lake Description	2004 Station Name	2012 Station Name	Zone	Easting	Northing
Main Basin of Snap Lake	PERI 4	SNAP LZ01	12V	507250	7053242
	-	SNAP LZ02	12V	508741	7053978
	PERI 7	SNAP LZ03	12V	508024	7051210
	-	SNAP LZ04	12V	509070	7050770
	PERI 8	SNAP LZ05	12V	509615	7053028
Northwest Arm of Snap Lake	PERI 3	SNAP LZ06	12V	503754	7053448
	PERI 1	SNAP LZ07	12V	502191	7052714
Northeast Lake	-	NEL LZ01	12V	508736	7059712
	-	NEL LZ02	12V	509921	7059851
	-	NEL LZ03	12V	511697	7058828

Note: Universal Transverse Mercator coordinates. Only a subset of the 2004 periphyton stations corresponded to the 2012 stations.

PERI = Periphyton sampling station; SNAP LZ = Snap Lake littoral sampling station; NEL LZ = Northeast Lake littoral sampling station; "-" = not sampled in 2004.



NORTHEAST LAKE



SNAP LAKE

1 0 1
SCALE KILOMETRES

LEGEND

- ★ LITTORAL STUDY STATION
- DEPTH CONTOUR (m)
- WATERCOURSE
- WATERBODY
- SNAP LAKE MINE FOOTPRINT

NORTHEAST LAKE REFERENCES

DIGITIZED FROM NTS TOPOGRAPHIC MAP 75 M/10 © 1985 HER MAJESTY THE QUEEN IN RIGHT OF CANADA. DEPARTMENT OF ENERGY, MINES AND RESOURCES. PROJECTION : TRANSVERSE MERCATOR, DATUM : NAD27, COORDINATE SYSTEM : UTM ZONE 12.

REFERENCE 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 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 PLANTSITE DRAWING OBTAINED FROM CLIENT. DATE: JUNE 12, 2012. "SNAPLAKE-08_STATMAP_2012". DRAWN BY Nampcy Solutions Ltd.

NOTES

MAP MOVED TO PROJECTION : TRANSVERSE MERCATOR, DATUM : NAD83, COORDINATE SYSTEM : UTM ZONE 12.

PROJECT
DE BEERS
GROUP OF COMPANIES

TITLE
LITTORAL SPECIAL STUDY MONITORING STATIONS IN SNAP LAKE AND NORTHEAST LAKE, 2012

Golder Associates

PROJECT	13-1349-0001.8200	FILE No. 13134900018200A12.1-1	
DESIGN	TD	2013-03-13	SCALE AS SHOWN REV. 0
CADD	JEF	2013-03-30	FIGURE 12.1-1
CHECK	LMB	2013-04-19	
REVIEW	PMC	2013-04-19	

12.1.2.2 Sampling Methods

Supporting Environmental Variables

Surface grab water samples were collected from the side of the boat from each littoral zone station for analyses of TP, total dissolved phosphorus (TDP), total nitrogen (TN), total dissolved nitrogen (TDN), DIC, and dissolved organic carbon (DOC). Field water quality parameters (water temperature, dissolved oxygen (DO), pH, and conductivity were measured at each littoral zone station using a YSI 600-QS multi-meter. Site-specific water quality data were required for the littoral zone study because AEMP water quality and plankton sampling stations are located further away in open-water areas. Because wave action results in good mixing of surface and deeper waters in the littoral zone of Snap Lake, surface water samples were considered representative of the water column in the shallow near-shore of the lake.

The TN and TP samples were collected directly in 250 millilitre (mL) pre-cleaned plastic bottles provided by the analytical laboratory. The TDP, TDN, DIC, and DOC samples were collected in two pre-cleaned 1 L plastic bottles. The TDP and TDN samples were then filtered through 0.45 µm glass fiber type C (GF/C) filters and the filtrate was collected in pre-labeled, pre-cleaned 250 mL plastic bottles. The DIC and DOC samples were filtered through Millipore cellulose nitrite filters, and the filtrate was collected in pre-labeled 250 mL ultra-clean plastic bottles. In addition, a field blank, using deionized water, was also prepared. All water chemistry samples were refrigerated at approximately 4 degrees Celsius (°C) before shipment to the University of Alberta Biogeochemical Analytical Services Laboratory (UofA) in Edmonton, Alberta, for analyses.

Epilithic Algae

Naturally occurring communities were assessed at a sampling depth of 2 m (i.e., below the wave-washed zone), where less of an affect from scouring occurs and within the euphotic zone, where light penetrates to the bottom. Epilithic algae samples were obtained from natural rock or boulders in areas of low slope (i.e., less than a 10 degree angle [°], assessed visually). They were removed from the rock surface by commercial divers using SCUBA-based techniques. The sampling technique used for this study is a widely accepted in-lake epilithic collection method (Turner et al. 1987) when performed by scientifically-trained divers, which unfortunately was not the case in 2012 but will be rectified in 2013 (Section 12.1.4). *In situ* scraping-brush samplers, based on a design created by Dr. Michael Turner and built by JS Micro Products (De Beers 2005), were used to scrape the epilithic algae from the rocks. These scrapers were designed to sample an area of 5 square centimetres (cm^2), while minimizing the amount of material that can be lost during sampling. Samples were collected following accepted protocols provided by Dr. Michael Turner, consistent with protocols used in 2004 (De Beers 2005), as outlined below.

At each of the ten stations, three composite samples, each consisting of five 5 cm^2 scrapings (sub-samples), were collected using 60 mL syringes (Figure 12.1-2 and Table 12.1-2). Sub-samples were collected within a 40 centimetre (cm) diameter rock area. If it was not possible to collect five sub-samples within this area (e.g., if rock angles exceeded 10°, or the rock area was too small), then sub-samples were collected from a suitable area within 1 m of the original 40 cm diameter area. After collection, the divers sealed the syringes and returned them to the surface. At the surface, the five syringes were combined to create a composite sample and placed into a pre-labeled Whirlpak™ bag. Samples were then transported and stored on ice for less than 12 hours (h) until sample preparation.

In addition to the scrapings, colour, adhesiveness, and thickness were visually assessed, and digital photographs of all sample locations were taken (Appendix 12A.2).

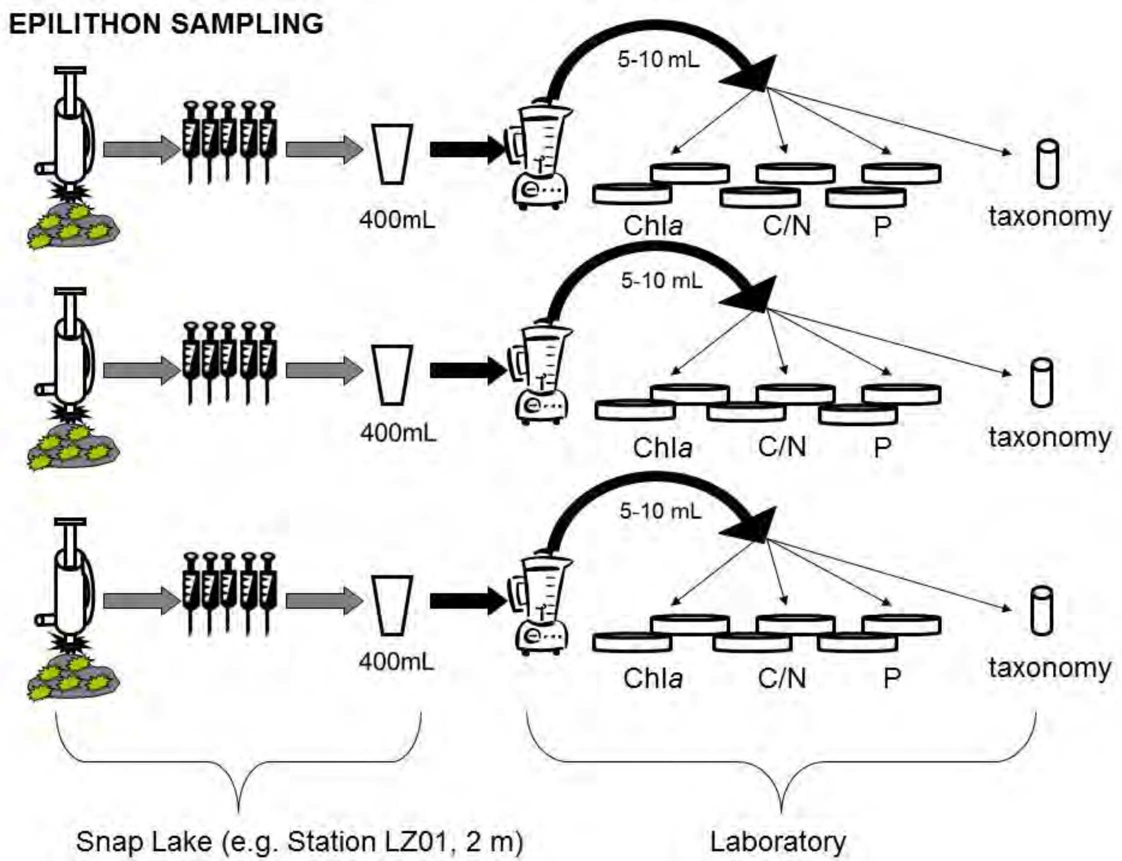
Table 12.1-2 Littoral Zone Sampling Program, 2012

Component	Depth	Analyses	Number of Stations (a)	Number of Samples per Monitoring Station	Duplicates	Total Number of Samples
Water Chemistry	Surface	Total N and P	10	1	No	10
		Dissolved N and P	10	1	No	10
		DIC and DOC	10	1	No	10
		QC Samples	1	-	-	1
Epilithic Algae including associated Bacteria and Detritus	2 m	Epilithic algal Community Composition and Biomass	10	3	No	30
		QC samples	-	3	-	3
		Chlorophyll *a*	10	3	Yes	60
		Particulate C/N	10	3	Yes	60
		Particulate P	10	3	Yes	60
Littoral Invertebrates	2 m	Littoral Invertebrate Community Composition	10	1	No	10

(a) Includes both Snap Lake and Northeast Lake samples.

"-" = not applicable; C = carbon, N = nitrogen; and P = phosphorus; DIC = dissolved inorganic carbon; DOC = dissolved organic carbon; m = metre; QC = quality control.

Figure 12.1-2 Overview of Epilithic Algae and Associated Bacteria and Detritus Sample Collection Methods



Note: Three samples were collected at each station and each sample was made up of five syringes.
mL - millilitre; Chla = Chlorophyll *a*; C = carbon; N = nitrogen; P = phosphorus.

Invertebrates

Littoral invertebrate sampling provided an initial qualitative assessment of those communities and provided information for selecting an appropriate sampling method. The purpose of sampling was to collect sufficient material to identify taxa present in the littoral zones of Snap Lake and Northeast Lake, and make field observations that would allow refining the sampling method. A comparison of two different sieve sizes (i.e., 250 and 500 micrometer [µm]) for collection was also undertaken (Section 12.1.2.3). Littoral invertebrates were collected at a depth of 2 m, after the epilithic algae sample collection, and approximately 50 m away from the epilithic algae sampling area to avoid disturbance.

One littoral invertebrate sample was collected at each station, for a total of ten stations (Table 12.1-2). An area that produced enough material for an approximately 100 mL sample volume from each station was sampled. Each station was swept with a coarse bristle broom to disturb the entire boulder area and detach the invertebrates from the boulder surface. Once sufficient material was suspended in the lake water, a 41 x 47 cm, 250 µm mesh net was swept through the water to catch the dislodged material. The suspended material was collected in the net, brought to the surface, emptied into a 500 mL plastic sample bottle, and preserved in 10% buffered formalin. Littoral invertebrate samples were sent to J. Zloty, Ph.D., in Summerland, British Columbia, for taxonomic analyses.

Field observations were also made regarding the presence of heavier, shelled or cased organisms that may not be efficiently collected by the sampling method used, thereby resulting in selectivity of the method toward small, light, easily-dislodged invertebrates. The information from the samples collected and field observations were considered relative to recommendations regarding sampling methods for future littoral work under the AEMP.

12.1.2.3 Sample Preparation, Sorting, and Taxonomic Identification

Epilithic Algae Sample Preparation

Sample preparation occurred less than 12 h after collection. The samples were stored on ice in the dark until preparation. Light level was set low during processing to avoid light damage to the chlorophyll samples. Each sample was prepared following protocols provided by Dr. Michael Turner and described below.

Each Whirlpak™ bag containing sample material was thoroughly mixed before transfer to a 500 mL graduated cylinder to measure the quantity of particulate matter (the settled volume) collected at each station. The contents of the graduated cylinder were then transferred to a household blender to homogenize the sample. The blender was set on the lowest speed and the slurry was blended for three one-second pulses. The resulting suspension was transferred to a 1 L stirring beaker, at a setting of seven on a Nuova II stirrer using a magnetic stir bar. Lake water, collected from the surface at the littoral zone sampling stations, was used to bring the final volume to 400 mL. Ten mL aliquots were removed using a large-bore syringe, and were filtered for duplicate chlorophyll *a* and particulate C, N, and P analyses. This volume of filtered sample was equivalent to 0.05 cm^2 of material on the filter. Subsamples were filtered through pre-ignited Whatman 25 mm GF/C (1.2 µm pore size) filters using a vacuum pump.

After filtering, the chlorophyll *a* and particulate C/N samples were desiccated for 12 to 24 h. The chlorophyll *a* samples were then wrapped in foil and frozen. The C/N samples were placed in a refrigerator following desiccation. The particulate P filters were not desiccated, but were kept in the refrigerator, prior to analysis. Chlorophyll *a* and particulate C, N, and P samples were kept on ice in coolers and shipped to the UofA for analyses.

The remaining suspension was used for epilithic algal community and biomass analyses. Subsamples of 20 mL volume were removed and transferred to 20 mL scintillation vials, where they were preserved with 4% acid Lugol's solution. Epilithic algal community composition and biomass samples were kept at room temperature and shipped in a cooler to D. Findlay, Plankton R Us, in Winnipeg, Manitoba, for analyses.

Epilithic Algal Taxonomic Identification

Subsamples of the preserved epilithic algal composites were analyzed using the modified Ütermohl technique (Nauwerck 1963). To break up detrital clumps, samples were sonicated at 20 kHz (Sonifer cell Disruptor, Model W140, Heat Systems, Ultrasonic Inc.) for up to two fifteen-second intervals, depending on the severity of the clumps. Two 2 mL sub-samples were allowed to settle for 24 h. Cells were identified to the lowest taxonomic unit using a phase-contrast inverted microscope at 125x and 400x magnification until a minimum of 100 cells of the dominant taxon were counted. Only viable cells that showed chloroplast presence were enumerated (Owen et al. 1978).

Algal taxonomy was based on taxonomic groupings by Hustedt (1930), Patrick and Reimer (1966), and Findlay and Kling (1979). In each sample, 50 cells of the most common taxa were measured by approximating cell shapes as geometric solids (Vollenweider 1974). For less common taxa, cells were measured as they were encountered, and estimates of cell size were based on less than 50 measurements. For simplicity, both algal cells and colonies are referred to in terms of algal cell size. Estimates of algal wet biomass were obtained from algal cell measurements assuming a specific gravity of 1 (Nauwerck 1963).

Invertebrate Taxonomic Identification

Samples were processed according to standard protocols based on recommendations in Gibbons et al. (1993) and Environment Canada (2002). Invertebrate samples were first washed through a 500 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 size. Because samples were generally small, laboratory subsampling was not required.

Invertebrates were identified to the lowest practical taxonomic level, typically genus, using recognized taxonomic keys (Soponis 1977; McAlpine et al. 1981; Wiederholm 1983; Oliver and Roussel 1983; 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. The 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.

Mesh Size Assessment

Samples from Northeast Lake and Snap Lake were collected using two different mesh sizes: a 250 µm mesh opening, and a 500 µm mesh opening. Organisms retained on these two mesh sieves were identified and enumerated separately to determine which of the two mesh sizes would collect the most representative sample of the invertebrate community.

12.1.2.4 Data Analyses

The Littoral Zone Special Study was designed to answer the key questions listed in Section 12.1.1.2. An overview of the analysis approach associated with each of the four key questions is provided in Table 12.1-3.

A qualitative review of data was completed. Summary statistics (i.e., sample size, arithmetic mean, median, minimum, maximum, standard deviation and standard error [SE]) were calculated for particulate C, N, P, chlorophyll *a*, and epilithic algal biomass and abundance.

Averages for the main basin of Snap Lake, northwest arm of Snap Lake, and Northeast Lake, were calculated for each supporting environmental variable: temperature, DO, conductivity, pH, and water column concentrations of TN, TP, TDN, TDP, DIC, and DOC. The percent of total lake area available for epilithic algal colonization was calculated using the following formula:

$$\text{Area of littoral zone (\%)} = \frac{\text{Area of the lake with depths from 0 to 4 m x 100}}{\text{Total lake surface area}} \qquad \text{[Equation 12-1]}$$

Molar ratios were calculated for each particulate nutrient parameter (i.e., C, N, and P). Particulate nutrient concentrations in micrograms per square centimetre ($\mu g/cm^2$) were converted to moles by dividing each nutrient by its respective molar mass to provide gram-atomic molar ratios. The ratios of carbon to nitrogen (C:N), carbon to phosphorus (C:P), and nitrogen to phosphorus (N:P) were then calculated.

The mean and SE for each molar ratio (C:N, C:P, and N:P) was calculated for each station. Ratios in the main basin and northwest arm of Snap Lake were compared to baseline ratios from 2004. The molar nutrient ratios were also compared to established values reported in the literature that indicate nutrient status and food quality (Healey and Hendzel 1980; Hillebrand and Sommer 1999; Elser et al. 2000).

The amount of carbon on the rock surface, in micromoles per square centimetre (µmol/cm^2), was divided by the chlorophyll *a* concentration (µg/cm^2) to produce a proportional estimate of chlorophyll-related changes in the system. The carbon to chlorophyll *a* ratio (C:chlorophyll *a*; µmol to µg) was used to determine nutrient status of the epilithic algae including associated bacteria and detritus (Healey and Hendzel 1980). The mean and SE of this ratio were calculated for each station. Ratios in the main basin and northwest arm of Snap Lake were compared to baseline ratios from 2004.

The percent algal carbon associated with the epilithic algae and associated bacteria and detritus was also calculated for each station, and values for Snap Lake were compared between 2012 and 2004. Percent algal carbon was calculated assuming 10% of the algal wet biomass was equal to algal dry weight. Half of the dry weight was assumed to be algal carbon, corresponding to 5% of the wet biomass (Frost et al. 2002). The estimated algal carbon was then divided by the measured particulate carbon (µg/cm^2) to estimate the proportion of viable algae.

Epilithic algal abundance and biomass data were divided into groups based on taxonomic results:

- Cyanobacteria;
- Chlorophyceae (chlorophytes);
- Bacillariophyceae (diatoms); and,
- “others” (i.e., dinoflagellates, chrysophytes, and euglenophytes).

The relative proportion accounted for by each group, based on both abundance and biomass, was calculated separately for each station, for 2004 and 2012. Stations in the main basin of Snap Lake were compared to those in the northwest arm of Snap Lake and Northeast Lake in 2012. In addition, relative abundances and biomass in Snap Lake were compared to baseline values from 2004.

Within-station variability in particulate C, N, P, chlorophyll *a*, and epilithic algal abundance and biomass was assessed by examining the coefficient of variation (CV) among samples for each station. Within-lake variability was characterized by examining station-to-station variability and spatial trends in Snap Lake and Northeast Lake.

The littoral invertebrate data were evaluated qualitatively (i.e., based on presence/absence) including a preliminary comparison of the littoral invertebrate community among the main basin of Snap Lake, northwest arm of Snap Lake, and Northeast Lake. The littoral invertebrate data were divided into major taxonomic groups, and densities of major invertebrate taxa were calculated for each station. In addition, since Diptera (mostly Chironomidae) dominated most samples at all stations, densities of each subfamily or tribe within the Chironomidae were plotted to compare sampling areas.

Invertebrate community structure was also examined by classifying the invertebrates at each station into functional feeding groups (filterers, collector-gatherers, scrapers, grazers, predators, and herbivores), using descriptions of feeding type for each taxon provided by Merritt and Cummins (1996). Non-insect taxa were assigned to feeding groups based on their general biology, and summaries in Pennak (1989) and Thorp and Covich (2001). When a taxon was assigned to two or more feeding groups, the number and percentage for that taxon in a sample were divided evenly among those functional feeding groups. Percentages of invertebrates in each functional feeding group were compared visually among sampling areas.

Table 12.1-3 Overview of Analysis Approach for Littoral Zone Special Study Key Questions, 2012

Key Question	Overview of Analysis Approach
• Can littoral monitoring be conducted in Snap Lake and Northeast Lake, and does the inherent variability in the littoral zone allow the detection of Mine-related changes?	This question will be answered after three years of the Littoral Zone Special Study. An annual assessment of the among-station and lake variability was done based on the 2012 data. The CV among the samples was calculated for each station. Variability among samples was examined in particulate C, N, P, ratios of C, N, and P, chlorophyll *a*, and epithic algal abundance and biomass. In addition, within lake variability was characterized by examining among-station variability and spatial trends in Snap Lake and Northeast Lake.
• What are the current ratios of particulate C:N, C:P, N:P, and C:chlorophyll *a*, and what is the current percent algal carbon in the littoral zones of the main basin of Snap Lake, the northwest arm of Snap Lake, and Northeast Lake? How do these values compare to baseline, and what do these values indicate about Mine-related changes in nutrient status and food quality for invertebrates and fish?	Summary statistics were calculated for particulate C, N, and P (Appendix 12A.3, Table 12A.3-5). The mean and SE were calculated for the molar ratios of C:N, C:P, N:P, C:chlorophyll *a*, and the percentage of algal carbon. These values were examined at each station in each lake; values from the main basin of Snap Lake were compared to values in the northwest arm of Snap Lake and Northeast Lake in 2012. Values in the main basin and northwest arm of Snap Lake were also compared to baseline (2004) values in Snap Lake. Nutrient ratios were also compared to values reported in the literature (Healey and Hendzel 1980; Hillebrand and Sommer 1999; Elser et al. 2000), which indicate nutrient status and food quality.
• What is the current status, in terms of relative abundance and relative biomass, of the epilithic algal communities in the main basin of Snap Lake, the northwest arm of Snap Lake, and Northeast Lake? Do these results provide any evidence of a Mine-related effect?	Summary statistics were calculated for total epilithic algal biomass and abundance. Mean relative abundance and biomass were calculated for each station, and stations in the main basin of Snap Lake were compared to those in the northwest arm of Snap Lake and Northeast Lake in 2012. Relative abundance and biomass in the main basin and northwest arm of Snap Lake were compared to baseline (2004) values in Snap Lake.
• What is the current invertebrate composition in the littoral zones of the main basin of Snap Lake, the northwest arm of Snap Lake, and Northeast Lake? Do these results provide any evidence of a Mine-related effect?	Relative densities of the major invertebrate taxa were calculated for each station and stations in the main basin of Snap Lake were compared to those in the northwest arm of Snap Lake and Northeast Lake in 2012.

C = carbon; CV = coefficient of variation; N = nitrogen; P = phosphorus.

12.1.3 Quality Assurance and Quality Control

Quality assurance (QA) and quality control (QC) procedures were applied during all aspects of the Littoral Zone Special Study so that the data collected were of acceptable quality. In accordance with Golder Associates Ltd. (Golder) QA/QC protocols, appropriate procedures were employed to support the collection of scientifically defensible and relevant data to address the objectives of this Special Study. The QA/QC protocols are designed so that field sampling, laboratory analyses, data entry, data analyses, and report preparation activities produce technically sound and scientifically defensible results.

The field QC program included collection of field blanks, replicates and split samples to assess potential sample contamination, and within-station variation and sampling precision. QC samples were submitted to the UofA and Plankton R Us for analyses. Field blank samples were submitted for water quality variables such as TP, TN, TDP, TDN, DIC, and DOC. Replicate and split samples were submitted for particulate nutrients (P, N and C) and chlorophyll *a* analyses, and an additional two epilithic algal split samples (approximately 10% of the total samples submitted) were submitted for re-count by the same taxonomist to verify counting efficiency. A full description of the QA/QC procedures is provided in Appendix 12A.1.

12.1.4 Summary of QA/QC Results

The methods used during this study to collect the littoral zone samples are widely accepted in-lake methods (Turner et al. 1987), when performed by scientifically-trained divers. Due to Golder and De Beers Health and Safety Procedures and dive restrictions in the Northwest Territories, only certified commercial divers were permitted to perform the SCUBA-based methods described by Turner et al. (1987) and Hille (2008). The certified commercial divers available for this work in 2012 were also doing other work for the Mine, and were not scientifically-trained. They were provided with an extensive briefing regarding the work that was required and how it should be conducted, including the importance of consistency in sampling. However, the diver doing the sampling was not consistent throughout the program, leading to potential sampler bias and field technique inaccuracies (Appendix 12A.1).

Since sampling methods were suboptimal in 2012, there is likely greater variability in the measurements of epilithic algal abundance, biomass, particulate nutrients and chlorophyll *a* than would be expected for a study of this nature. The resulting uncertainty diminishes the robustness of the data analyses and the ability to detect Mine-related changes; however, it does not negate the data. To reduce the potential effects of suboptimal sampling methods, data were analyzed proportionally (i.e., as relative percent abundance and biomass, and as molar nutrient ratios of the particulate C: N, C: P, N: P and C:chlorophyll *a*) for both the 2004 and 2012 data. The use of proportional data allows for comparisons to be made among stations, between lakes and years (Appendix 12A.1).

Issues with the sampling technique may have also caused an under-representation of the cyanobacteria by biasing the samples towards firmly attached algae rather than light, flocculent forms, which were observed in the photographs (Appendix 12A.1). Collections performed by scientifically-trained divers, during the 2013 program, will resolve this potential issue.

A species-level presence or absence assessment of the community was not performed, because different taxonomists analyzed the samples in 2004 and 2012. This is not expected to influence comparisons of epilithic algal communities based on major taxonomic groups. An investigation comparing the taxonomic identifications from the two separate taxonomists will occur in 2013, and a full species-level assessment will be provided following the three years of this Special Study, in 2015.

Based on results of the QC for field blanks and split samples, potential issues were identified for particulate C, N, P, and chlorophyll *a* (Appendix 12A.1). While none of the field blank results from 2012 exceeded the QC criterion, two replicate values showed signs of contamination and were therefore removed from further analyses: total dissolved N at SNAP LZ01; and, DIC at SNAP LZ05. These data were retained in Appendix 12A.3, Table 12A.3-1, but an X qualifier was applied to the table to indicate that the data were removed before analyses. Of the split samples, 31 (34% of total) results showed a relative difference of more than 20%, resulting in QC flags. The majority of flagged split samples occurred in Northeast Lake: 8 of 10 QC flags for particulate C; 6 of 11 QC flags for particulate P; and, 8 of 10 QC flags for particulate N. Overall, the QC results of the taxonomy data indicated that the 2012 littoral zone data were of acceptable quality and no data were invalidated. Detailed QA/QC results are provided in Appendix 12A.1.

12.1.5 Results

Results are presented starting with supporting environmental variables consisting of open-water nutrient concentrations, littoral zone nutrient concentrations, and nutrient ratios. Epilithic algal community variables (biomass, abundance, and relative proportions of each) are examined next, followed by a qualitative review of the littoral zone invertebrate community. Appendices 12A.1, 12A.2 and 12A-3 contain detailed results from all components of the study, as follows:

- Appendix 12A.1 Littoral Zone Quality Assurance and Quality Control;
- Appendix 12A.2 Littoral Zone Photographs;
- Table 12A.3-1 Water Quality Data;
- Table 12A.3-2 Particulate Nutrients and Chlorophyll *a*;
- Table 12A.3-3 Epilithic Algal Abundance and Biomass Data;
- Table 12A.3-4 Littoral Zone Invertebrate Data; and,
- Table 12A.3-5 Summary Statistics/Variability Among Replicates.

12.1.5.1 Supporting Environmental Variables

Snap Lake and Northeast Lake are small, shallow Arctic lakes (Table 12.1-4). In 2012, neither lake stratified thermally during the open-water period. Snap Lake is usually ice-covered by mid- to late October, and ice-free by early to mid-June. There were 140 days of open-water in 2012 (Section 1). Information on the number of days of ice-cover for Northeast Lake is not available; however, the number of open-water days in Northeast Lake is likely comparable to Snap Lake because the two lakes are close to one another and of similar size.

Northeast Lake (1,843 hectares [ha]) has a slightly larger surface area compared to Snap Lake (1,566 ha; Table 12.1-4). Within Snap Lake, the main basin (1,202 ha) is larger than the northwest arm (364 ha). Both lakes contain relatively large littoral zones due to their low sloping and rocky shorelines. These littoral zones consist of various substrata available for algal colonization (e.g., rock shelves, large boulders, organic sediment, and gravel). The abundance of surfaces available for colonization and good light penetration (Secchi depths greater than 6 m; Section 5) in these lakes provide optimal conditions for abundant epilithic algal growth (Wetzel 2001). The percent of the lake area available for epilithic algal colonization within the northwest arm of Snap Lake (59%) was greater than in the main basin of Snap Lake (48%), which was greater than that available in Northeast Lake (40%).

At the time of sampling in mid-August 2012, there was little variation in temperature and dissolved oxygen (DO) among the three sampling areas (Table 12.1-5). Differences were noted in pH; estimates for pH were high at two stations, SNAP LZ01 (8.1) and SNAP LZ02 (7.9), in the main basin of Snap Lake. Estimates at the other stations (SNAP LZ03 to LZ05) were similar (7.5) to those observed in the northwest arm of Snap Lake (7.4) and Northeast Lake (7.3).

Concentrations of DOC ranged from 2,350 micrograms per litre (µg/L) in the northwest arm of Snap Lake to 4,640 µg/L in the main basin of Snap Lake (Table 12.1-5). The lowest concentrations of DOC were observed in Northeast Lake (1,733 µg/L). Concentrations of DIC were highest in the northwest arm of Snap Lake (4,350 µg/L), followed by the main basin of Snap Lake (3,575 µg/L), and Northeast Lake (3,400 µg/L).

These lakes have naturally low nutrient concentrations, which do not support abundant phytoplankton populations (De Beers 2002). Water column concentrations of TN, TP, TDN, and TDP were lower in Northeast Lake compared to Snap Lake in 2012 (Table 12.1-5). Whole-lake mean TN concentration was greater in the main basin of Snap Lake (2,407 µg/L) compared to the northwest arm (2,350 µg/L) and Northeast Lake (164 µg/L), which reflects the input of the Mine discharge to Snap Lake; TDN closely followed the same pattern as TN, because a large fraction of the TN was TDN. Mean TP concentration was lower in the main basin of Snap Lake and Northeast Lake (2.0 and 2.3 µg/L, respectively) compared to the northwest arm of Snap Lake (4.0 µg/L) in 2012. In addition, mean TP concentration in the main basin of Snap Lake in 2012 was lower than reported in 2004 (3.2 µg/L). Open-water nutrient concentrations collected as part of the plankton component (Section 5) show that both Northeast Lake and Snap Lake are phosphorus limited systems.

Table 12.1-4 Physical Characteristics of Snap Lake and Northeast Lake

Parameter	Units	Snap Lake		Northeast Lake
		Main Basin	Northwest Arm	
Surface area	ha	1,202	364	1,843
Area of Littoral zone [(a)]	%	47.7	59.2	40.0

(a) Area of Littoral Zone (%) = (Area of lake with depths from 0 to 4 m / Total lake surface area) x 100.
ha = hectares;% = percent.

Table 12.1-5 Water Chemistry at the Littoral Zone Sampling Stations in Snap Lake and Northeast Lake in 2004 and 2012

Parameter	Units	Snap Lake 2004 [(a)]	Snap Lake 2012 [(b)]		Northeast Lake 2012
			Main Basin	Northwest Arm	
Mean Temperature	°C	-	16.6	15.6	16.2
Mean Dissolved Oxygen	mg/L	-	9.9	10.4	9.8
Mean Conductivity	µS/cm	26	397	125	22
Mean pH	-	7.0	7.7	7.4	7.3
Mean Dissolved Inorganic Carbon	µg/L	-	3,575	4,350	3,400
	µmoles/L [(c)]	-	280	363	283
Mean Dissolved Organic Carbon	µg/L	-	4,640	2,350	1,733
	µmoles/L	-	387	196	144
Mean Total Nitrogen	µg/L	1,524	2,407	426	164
	µmoles/L	109	172	30	12
Mean Total Phosphorus	µg/L	3.22	2.0	4.0	2.3
	µmoles/L	0.10	0.06	0.13	0.07
Mean Total Dissolved Nitrogen	µg/L	1,318	1,920	344	174
	µmoles/L	94	137	25	12
Mean Total Dissolved Phosphorus	µg/L	-	1.0	1.0	1.0
	µmoles/L	-	0.03	0.03	0.03

(a) 2004 information based on the annual open-water whole lake average data provided by the water quality component (De Beers 2005).
(b) 2012 information is based on surface water collected during the August program at littoral zone sampling stations.
(c) Molar concentrations were calculated by dividing the nutrient concentrations by their respective atomic values.
°C = degrees Celsius; mg/L = milligrams per litre; µg/L = micrograms per litre; µmoles/L = micromoles per litre; µS/cm = microSiemens per centimetre; "-" = data unavailable or not measured.

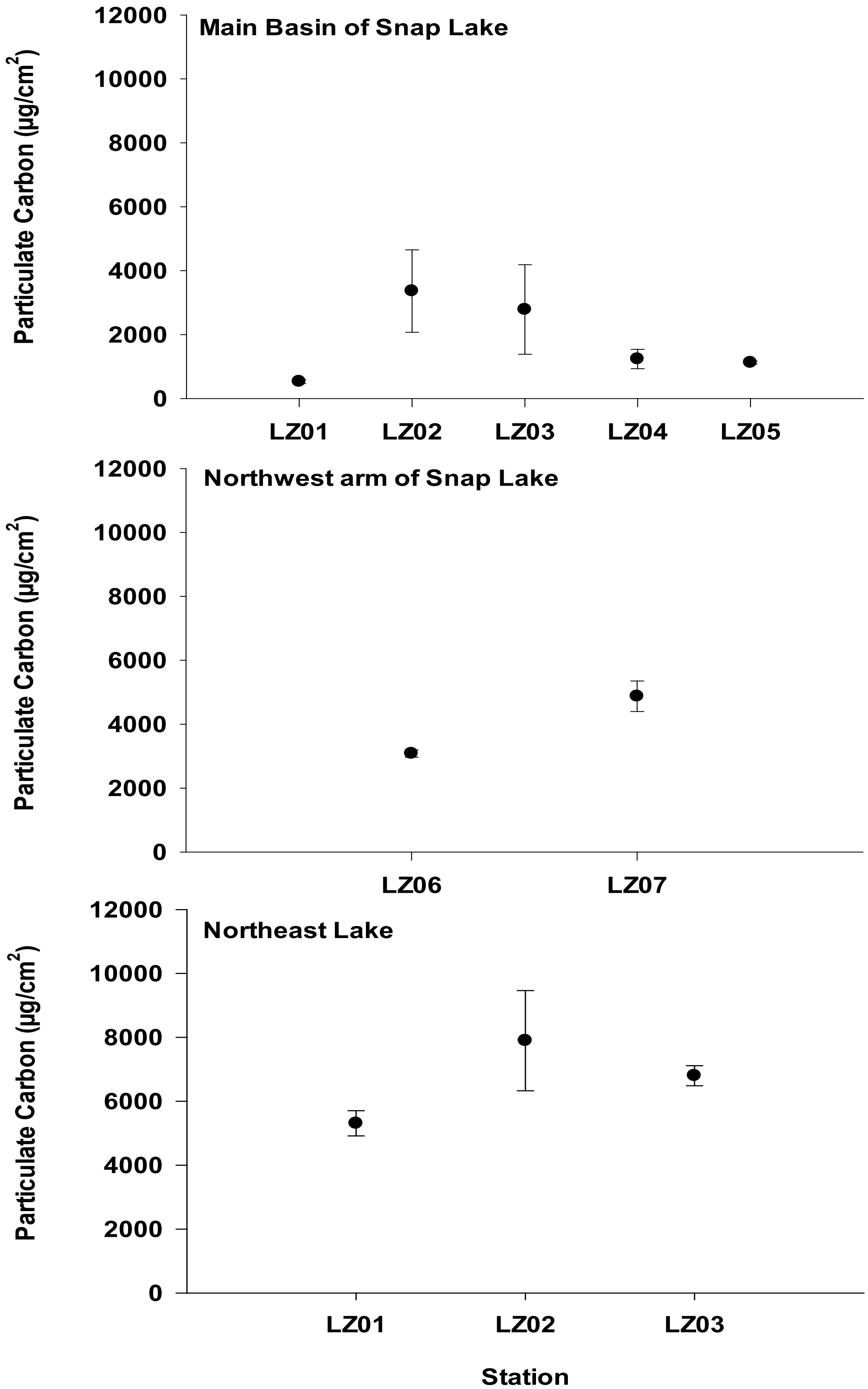
12.1.5.2 Littoral Zone Nutrients

Particulate nutrients in the epilithic algae and associated bacteria and detritus tended to be higher in Northeast Lake and the northwest arm of Snap Lake compared to the main basin (Figures 12.1-3 to 12.1-5). Lower mean particulate carbon concentrations (532 to 3,362 µg/cm^2) were observed in the main basin of Snap Lake compared to the northwest arm (3,082 to 4,874 µg/cm^2) and Northeast Lake (5,309 to 7,898 µg/cm^2).

The lowest particulate carbon concentrations were observed at SNAP LZ01 (532 µg/cm^2), while the highest particulate carbon concentrations were observed at NEL LZ02 (7,898 µg/cm^2) Particulate phosphorus concentrations in the main basin littoral zone ranged from 2 to 12 µg/cm^2 in 2012 (Figure 12.1-4). Concentrations in the northwest arm of Snap Lake ranged from 9 to 41 µg/cm^2, while concentrations in Northeast Lake ranged from 9 to 20 µg/cm^2. Particulate nitrogen concentrations in the main basin littoral zone ranged from 45 to 553 µg/cm^2 in 2012 (Figure 12.1-5). Concentrations in the northwest arm ranged from 293 to 844 µg/cm^2, while concentrations in Northeast Lake ranged from 331 to 614 µg/cm^2. Overall, lower particulate P and N concentrations were observed in the epilithic algae and associated bacteria and detritus in the main basin of Snap Lake compared to the northwest arm and Northeast Lake.

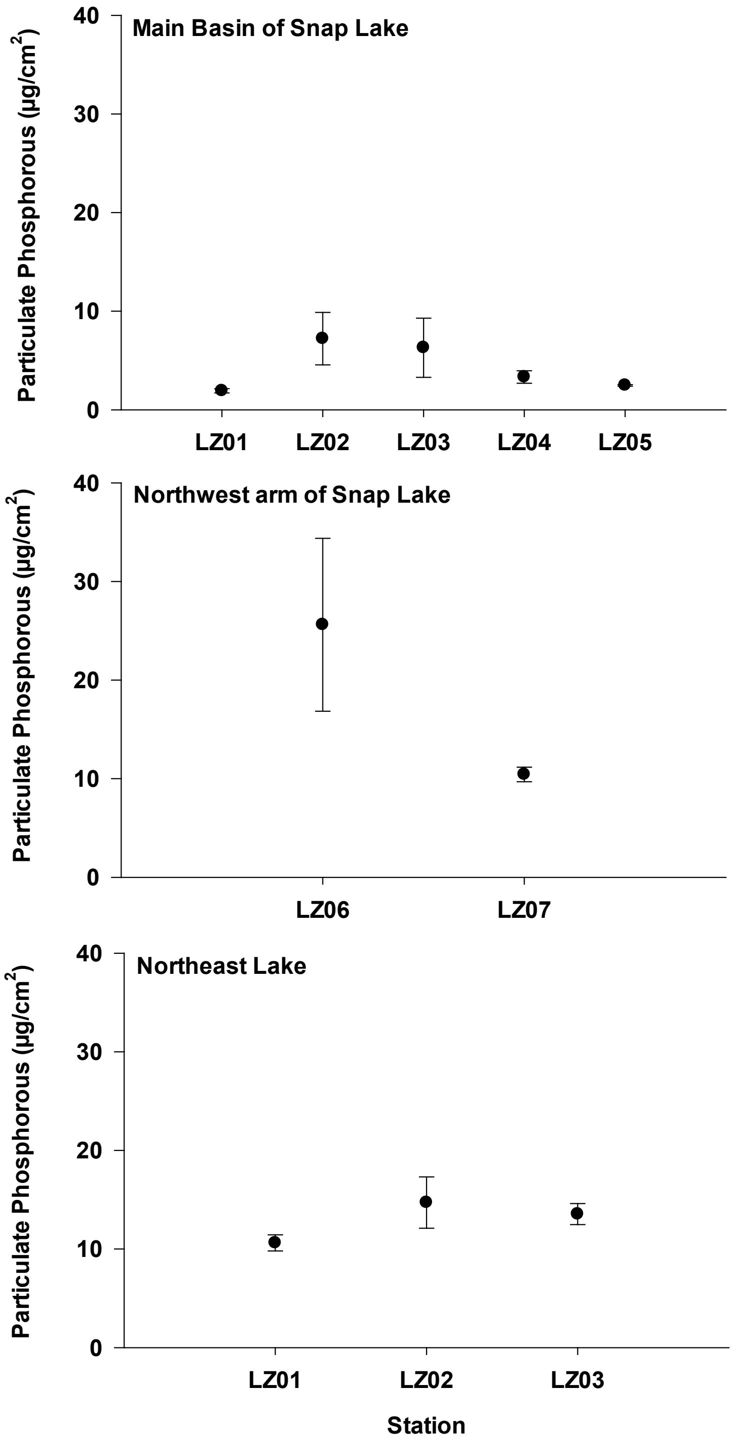
Spatial patterns were similar among the three particulate nutrients measured (Figures 12.1-3 to 12.1-5). The northwest arm of Snap Lake was an exception, where particulate C was higher at SNAP LZ07 than at SNAP LZ06 and particulate P and N were higher at SNAP LZ06 compared to SNAP LZ07. Overall, within-station variability was higher at SNAP LZ02 and SNAP LZ03 in the main basin of Snap Lake and SNAP LZ06 (except for carbon) in the northwest arm of Snap Lake. In addition, within-station variability was higher at NEL LZ02 in Northeast Lake compared to all other littoral zone stations in Snap Lake or Northeast Lake.

Figure 12.1-3 Spatial Trends in Particulate Carbon (µg/cm²), Epilithic Algae and Associated Bacteria and Detritus, Snap Lake and Northeast Lake, August 2012



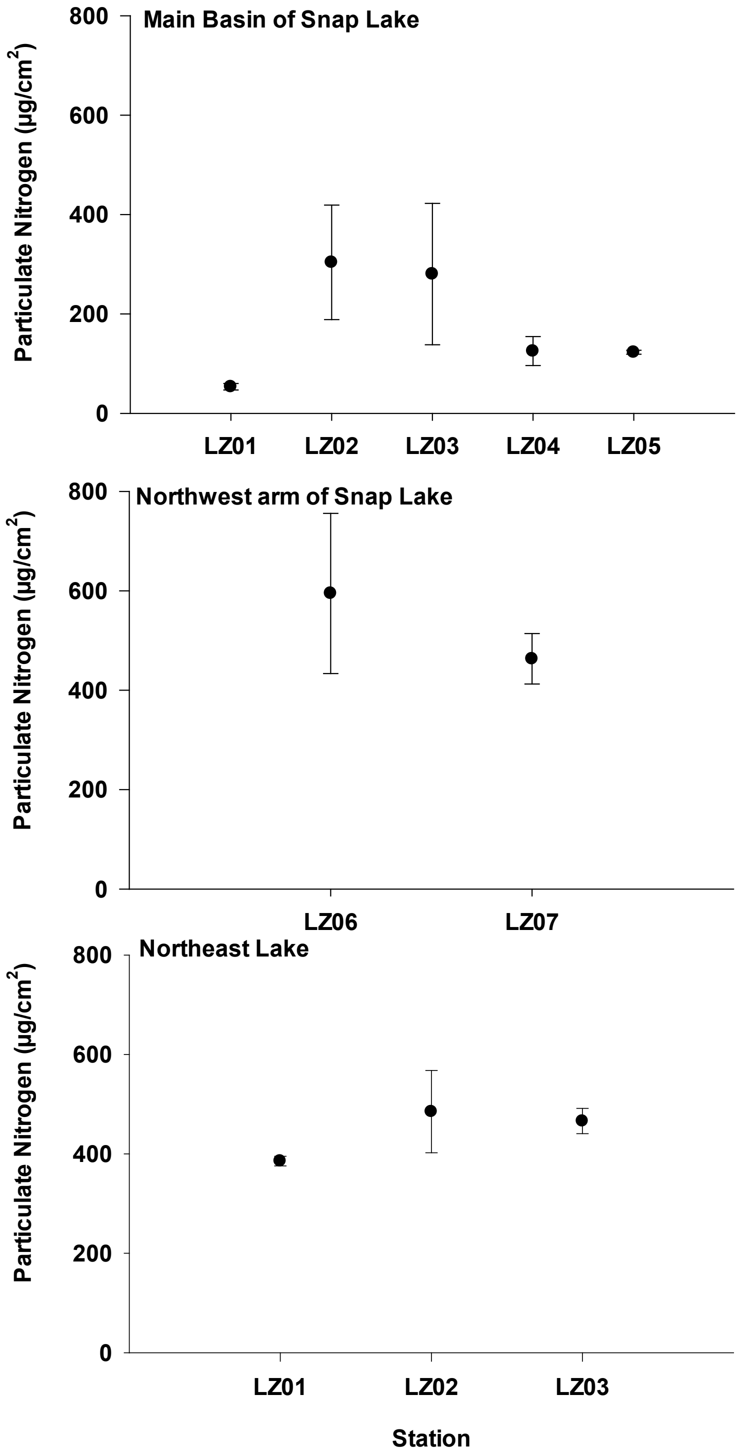
Note: Error bars represent standard error of the mean; µg/cm² = micrograms per square centimeter.

Figure 12.1-4 Spatial Trends in Particulate Phosphorus (µg/cm²), Epilithic Algae and Associated Bacteria and Detritus, Snap Lake and Northeast Lake, August 2012



Note: Error bars represent standard error of the mean; µg/cm² = micrograms per square centimeter.

Figure 12.1-5 Spatial Trends in Particulate Nitrogen (µg/cm²), Epilithic Algae and Associated Bacteria and Detritus, Snap Lake and Northeast Lake, August 2012



Note: Error bars represent standard error of the mean; µg/cm² = micrograms per square centimetre.

Epilithic Algae Nutrient Molar Ratios

C:N ratio

The C:N molar ratio decreased from 2004 to 2012 at all stations in Snap Lake (Figure 12.1-6). The C:N molar ratio ranged from 14 to 18 in 2004, while in 2012 the C:N molar ratio ranged from 4 to 13. Molar ratios were higher in Northeast Lake, ranging from 14 to 23.

Differences in the ratio of C:N were observed between the two stations in the northwest arm. The C:N molar ratios measured within each of the samples at SNAP LZ07 were similar (ranging from 11 to 13) to values observed in the main basin of Snap Lake (4 to 13). The C:N nutrient molar ratios measured within each of the samples at SNAP LZ06 were highly variable. Of the three samples at SNAP LZ06, one was removed because of QC issues associated with the particulate C values (Appendix 12A.1). Similar to the main basin of Snap Lake, the C:N molar ratios in Northeast Lake did not show a clear spatial pattern. However, variability around each mean value was greater in Northeast Lake, particularly at NEL LZ01 and NEL LZ02, compared to the main basin of Snap Lake, indicating greater within-station variability in the nutrient ratios in Northeast Lake.

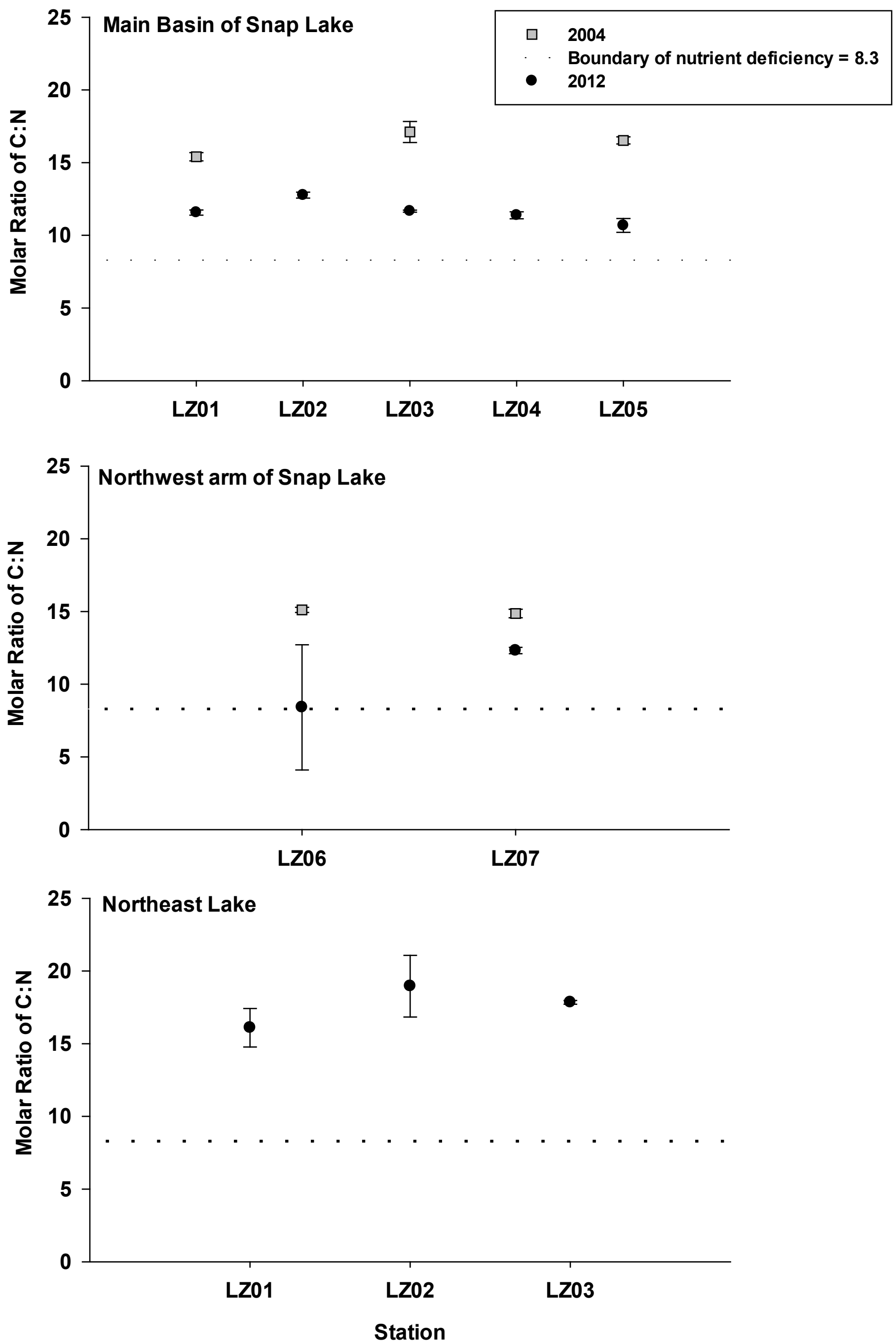
C:P Ratio

The average C:P molar ratios at SNAP LZ01 and SNAP LZ05 in the main basin and SNAP LZ06 in the northwest arm of Snap Lake were similar in 2004 and 2012 (Figure 12.1-7). With the exception of SNAP LZ01, C:P molar ratios were similar at all stations in the main basin of Snap Lake, ranging from 931 to 1,172. C:P molar ratios at all stations in Northeast Lake were also similar, ranging from 1,210 to 1,397. The mean C:P molar ratio at SNAP LZ03 in the main basin decreased from 2004 to 2012. The opposite trend was observed at SNAP LZ07 in the northwest arm of Snap Lake; the mean C:P molar ratio at SNAP LZ07 increased from 2004 to 2012.

N:P Ratio

The N:P nutrient molar ratios in Northeast Lake and in the main basin of Snap Lake were similar; however, station to station variability within the main basin of Snap Lake (ratios ranging from 48 to 111) was greater than in Northeast Lake (ratios ranging from 67 to 97) in 2012 (Figure 12.1-8). Average N:P molar ratios were similar between 2004 and 2012, with the exception of SNAP LZ07, where ratios were higher in 2012.

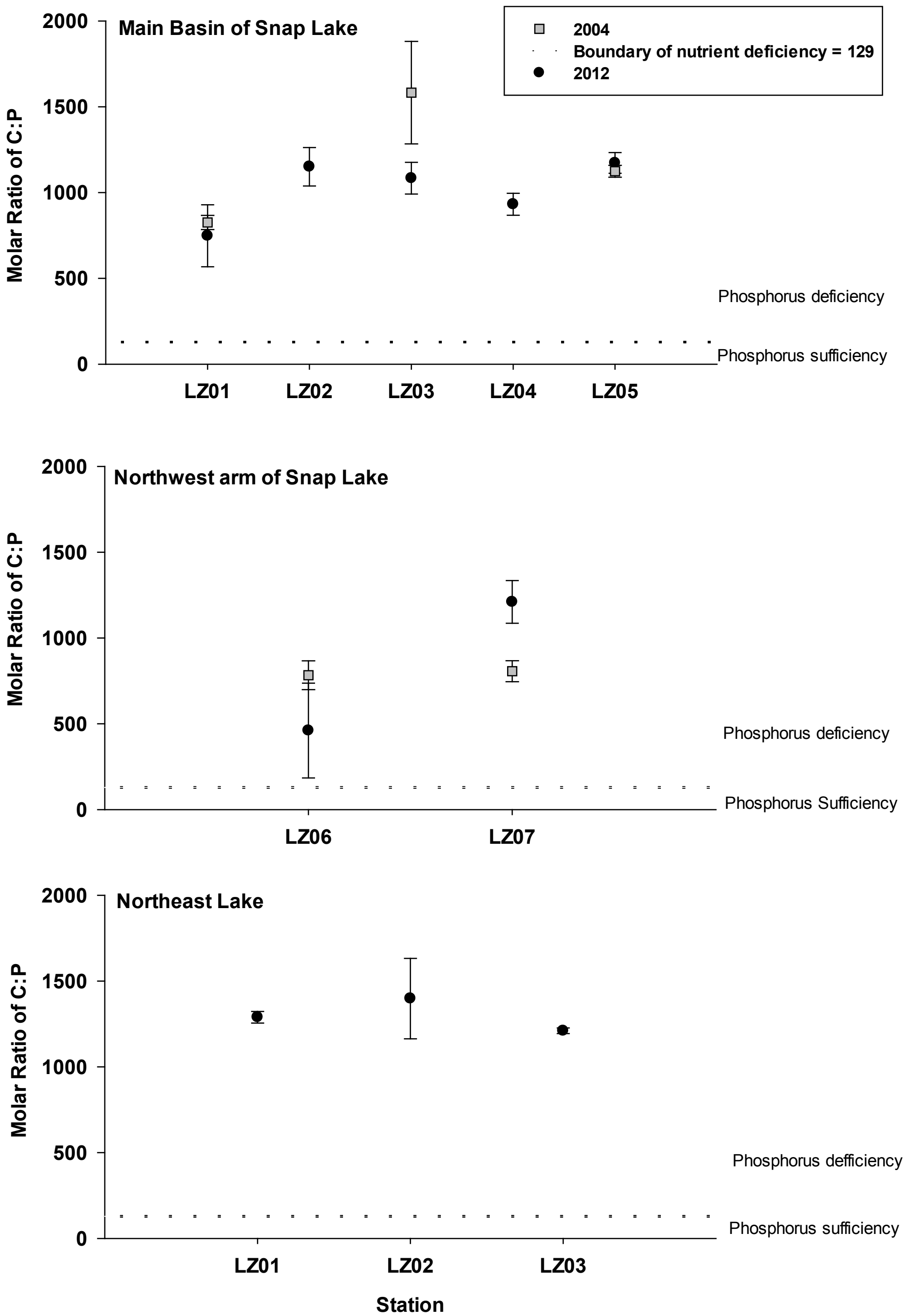
Figure 12.1-6 Spatial Trends in Molar Ratios of Carbon to Nitrogen, Epilithic Algae and Associated Bacteria and Detritus, Snap Lake 2004 and 2012, Northeast Lake 2012



Note: Error bars represent standard error of the mean; boundary of nutrient deficiency based on Healey and Hendzel 1980.

C= carbon; N = nitrogen.

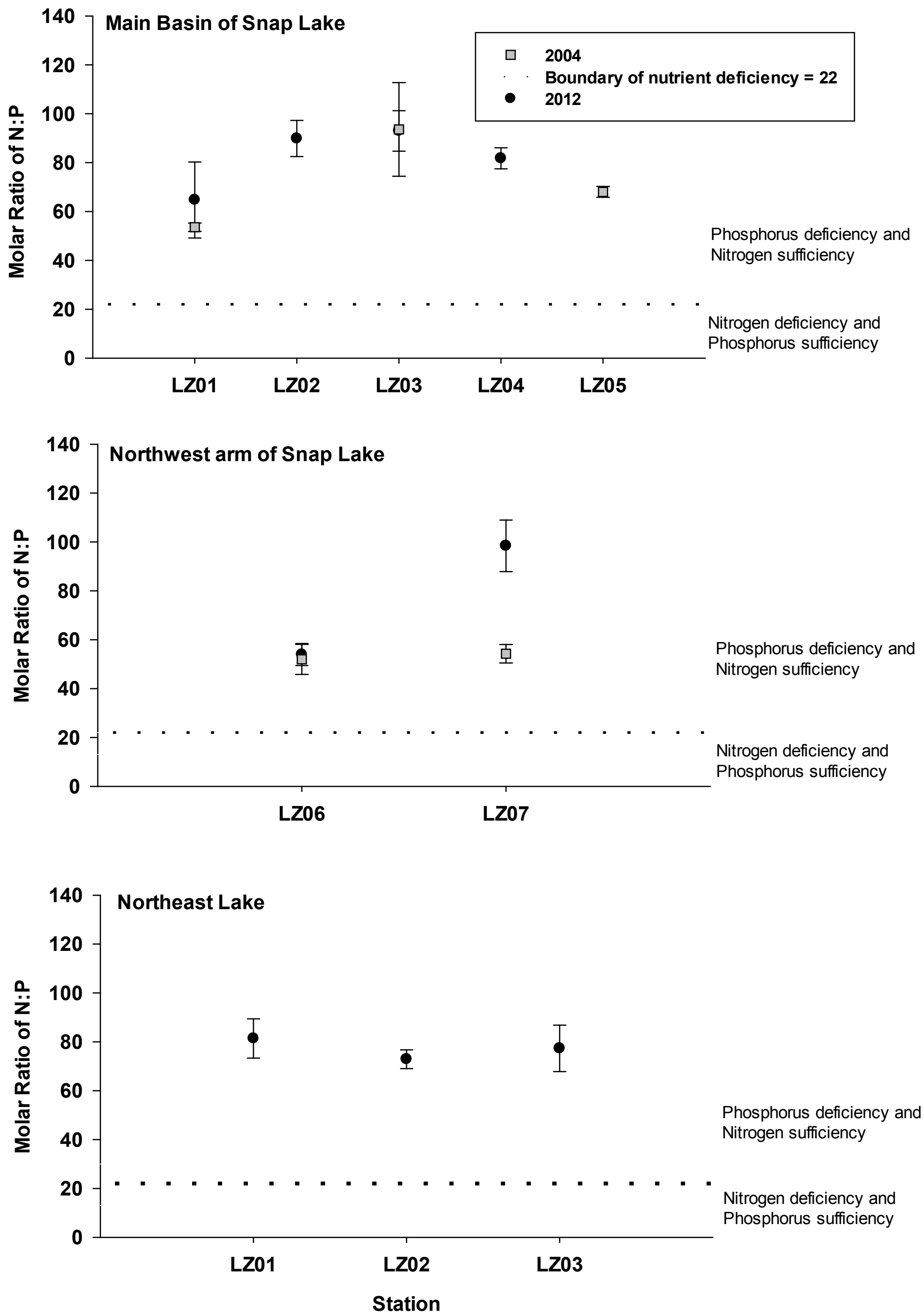
Figure 12.1-7 Spatial Trends in Molar Ratios of Carbon to Phosphorus, Epilithic Algae and Associated Bacteria and Detritus, Snap Lake 2004 and 2012, Northeast Lake 2012



Note: Error bars represent standard error of the mean; boundary of nutrient deficiency based on Healey and Hendzel 1980.

C= carbon; P = phosphorus.

Figure 12.1-8 Spatial Trends in Molar Ratios of Nitrogen to Phosphorus, Epilithic Bacteria and Associated Bacteria and Detritus, Snap Lake 2004 and 2012, Northeast Lake 2012



Note: Error bars represent standard error of the mean; boundary of nutrient deficiency based on Hillebrand and Sommer 1999.

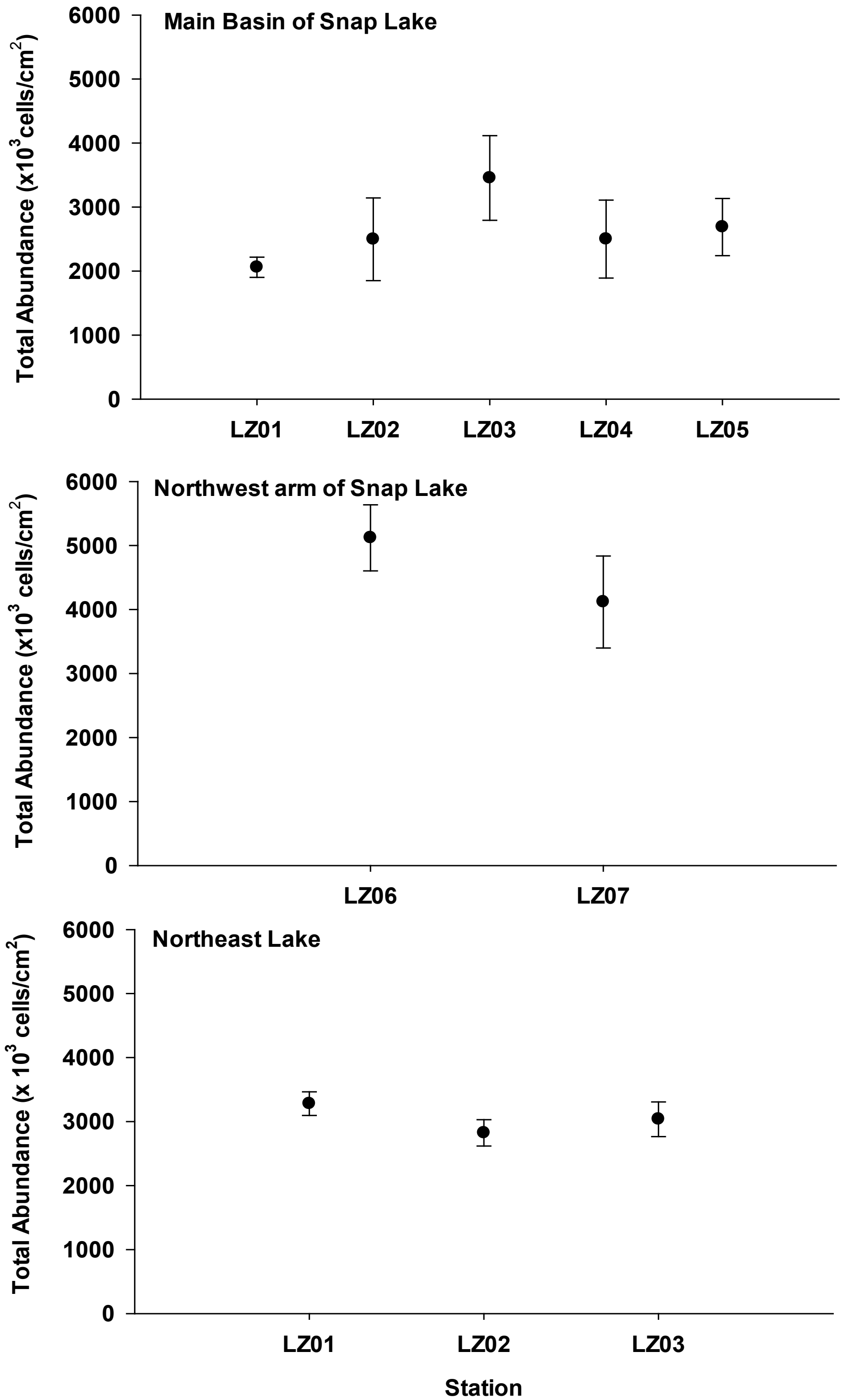
N= nitrogen; P = phosphorus.

12.1.5.3 Epilithic Algal Abundance and Biomass

Epilithic algal abundance in the main basin of Snap Lake in 2012 ranged from 1,494,401 to 4,547,804 cells per square centimetre (cells/cm^2). These values were similar to abundances observed in Northeast Lake, where abundance ranged from 2,476,314 to 3,617,571 cells/cm^2 (Figure 12.1-9). The lowest mean epilithic algal abundance was observed at SNAP LZ01 in the main basin of Snap Lake (2,060,006 cells/cm^2) and the highest abundance was observed at SNAP LZ06 in the northwest arm (5,120,107 cells/cm^2). Overall, variability in Snap Lake tended to be high compared to Northeast Lake.

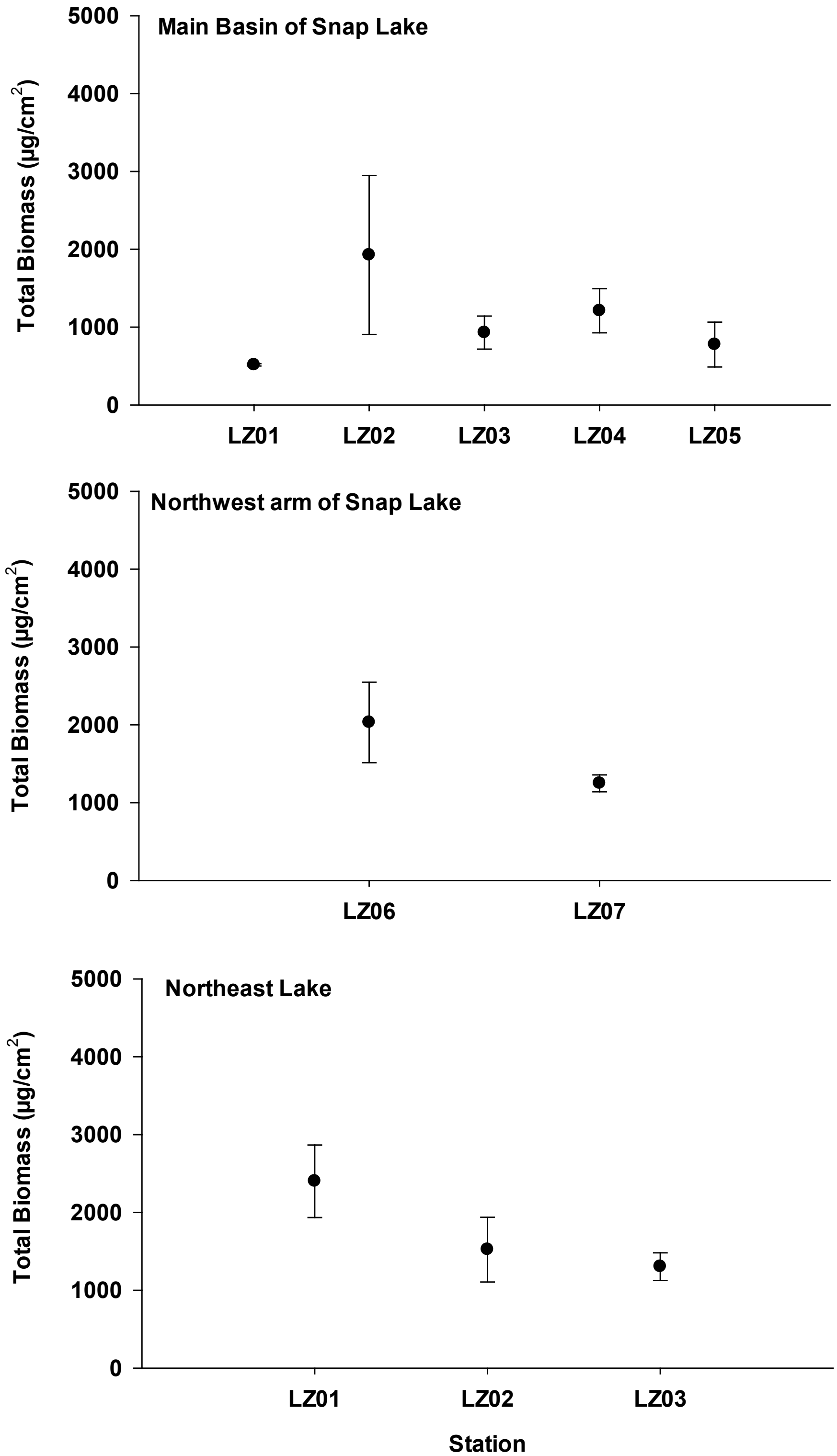
In the main basin of Snap Lake, epilithic algal biomass ranged from 306 to 1,585 μg/cm^2 in 2012, with the exception of some higher values at SNAP LZ02. Biomass in Snap Lake was lower compared to Northeast Lake, where it ranged from 980 to 3,307 μg/cm^2 (Figure 12.1-10). Biomass at SNAP LZ02 in the main basin of Snap Lake was higher and more variable than at the other main basin stations, ranging from 429 to 3,878 μg/cm^2. The lowest mean biomass was observed at SNAP LZ01 (515 μg/cm^2), which is the station closest to the diffuser. Biomass estimates at stations in Northeast Lake were similar to stations in the northwest arm of Snap Lake, and ranged from 1,040 to 2,786 μg/cm^2. Overall, variability associated with epilithic algal biomass tended to be high in both lakes.

Figure 12.1-9 Epilithic Algal Abundance in Snap Lake and Northeast Lake, August 2012



Note: Error bars represent standard error of the mean.
cells/cm^2 = cells per square centimetre.

Figure 12.1-10 Epilithic Algal Biomass in Snap Lake and Northeast Lake, August 2012



Note: Error bars represent standard error of the mean.
µg/cm² = micrograms per square centimeter.

12.1.5.4 Volume of Particulate Matter and Percent Viable Algae

Particulate matter, measured as settled volume, was lower in the main basin of Snap Lake than in the northwest arm and Northeast Lake (Table 12.1-6). Settled volume in the main basin of Snap Lake ranged from 3 to 18 mL. Higher settled volumes were observed at SNAP LZ02 and SNAP LZ03, compared to the other stations in the main basin. The lowest settled volume was observed at station SNAP LZ01, the station closest to the diffuser. Variation was high in the settled volume estimates of particulate matter between the two northwest arm stations; SNAP LZ06 had settled volumes ranging from 35 to 60 mL, while SNAP LZ07 had settled volumes ranging from 10 to 25 mL. Settled volumes in Northeast Lake were less variable, with values ranging from 47 to 50 mL among the three stations.

Table 12.1-6 Settled Volume of Particulate Matter in Snap Lake and Northeast Lake in August 2012

Lake	Station	Sample A (mL)	Sample B (mL)	Sample C (mL)	Station Mean (mL)
Main Basin of Snap Lake	SNAP LZ01	3	3	3	3
	SNAP LZ02	25	25	4	18
	SNAP LZ03	1	20	10	10
	SNAP LZ04	5	10	2	6
	SNAP LZ05	-	-	-	-
Northwest Arm of Snap Lake	SNAP LZ06	35	45	60	47
	SNAP LZ07	10	15	25	17
Northeast Lake	NEL LZ01	40	50	50	47
	NEL LZ02	50	50	40	47
	NEL LZ03	50	50	-	50

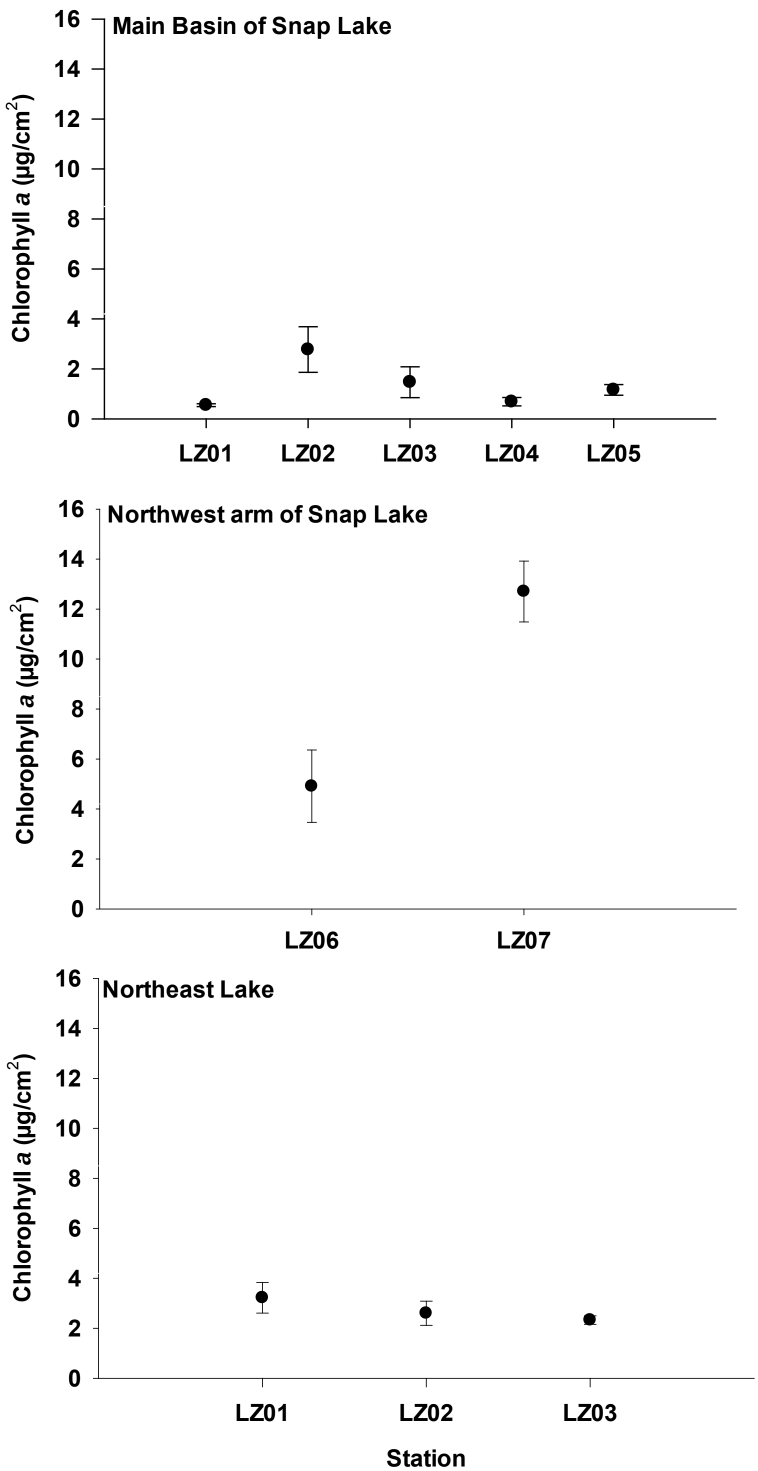
mL = millilitres; "-" = no data; sample lost by divers.

Chlorophyll *a* concentrations were similar between the main basin of Snap Lake and Northeast Lake (i.e., less than 3.2 µg/cm^2; Figure 12.1-11). Higher mean chlorophyll *a* concentrations were observed in the northwest arm of Snap Lake.

Northeast Lake had higher C:chlorophyll *a* ratios, ranging from 105 to 408, compared to ratios observed in Snap Lake, which ranged from 9 to 220 (Figure 12.1-12). Lower C:chlorophyll *a* ratios were also observed at stations in the northwest arm (9 to 77) compared to the main basin of Snap Lake (63 to 220). Ratios of C:chlorophyll *a* were similar between 2004 and 2012, with the exception of SNAP LZ07 in the northwest arm, which showed a decreased between 2004 and 2012.

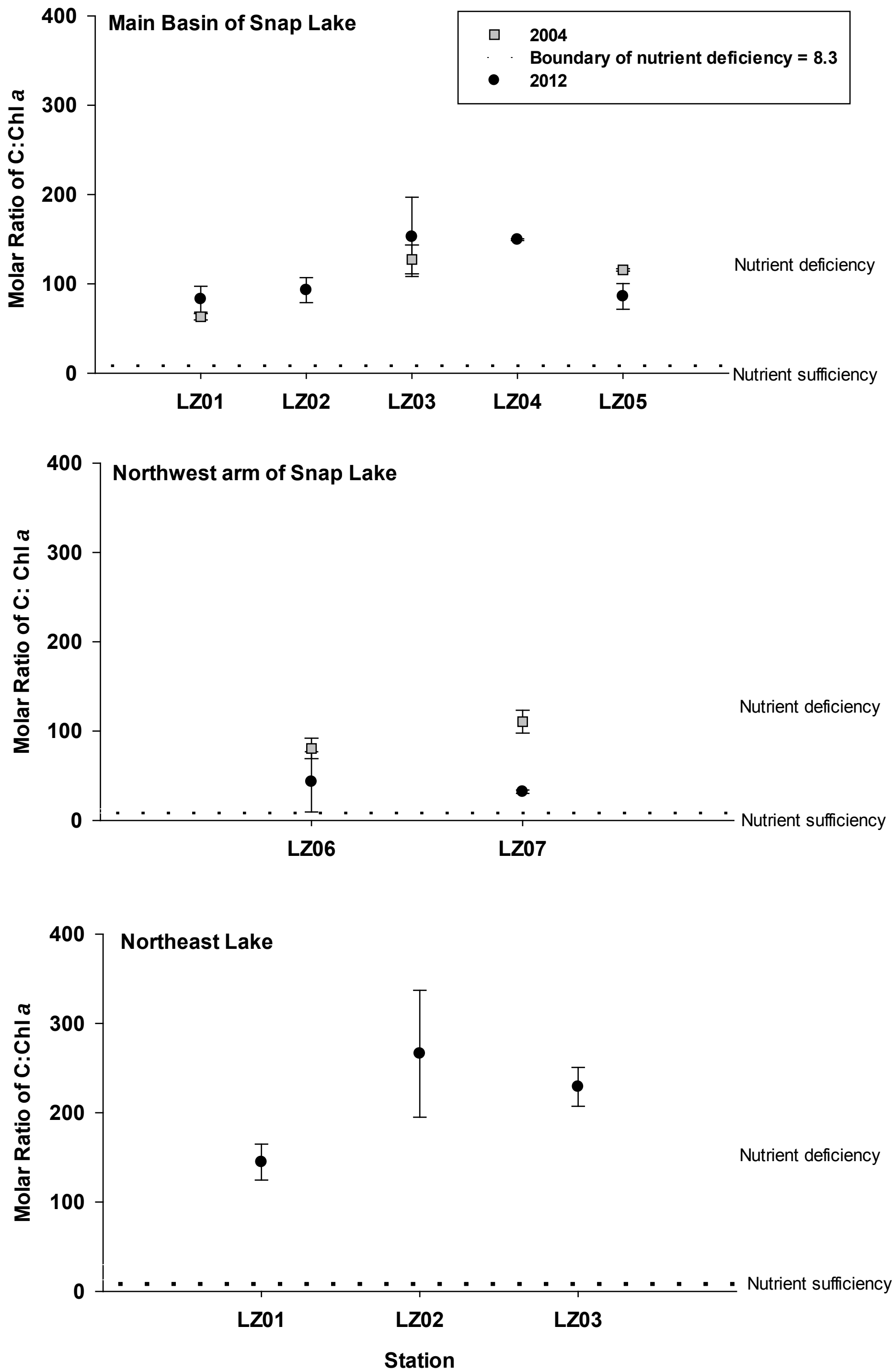
The percentage of viable algal carbon has increased in the main basin of Snap Lake since 2004, but remained similar between 2004 and 2012 in the northwest arm of Snap Lake (Figure 12.1-13). The highest average value of algal carbon (5%) was found at SNAP LZ01 in the main basin of Snap Lake, which is the station closest to the diffuser. The lowest average value of algal carbon (1.3%) was found at SNAP LZ07 in the northwest arm, which is the furthest station from the diffuser. Values were similar between the northwest arm of Snap Lake and Northeast Lake, and ranged from 0.9% to 2.7%. In contrast, values in the main basin of Snap Lake ranged from 2.3% to 5%.

Figure 12.1-11 Temporal Trends in Epilithic Algae Chlorophyll *a* Concentrations, Snap Lake and Northeast Lake, August 2012



Note: Error bars represent standard error of the means.
µg/cm^2 = micrograms per square centimetre.

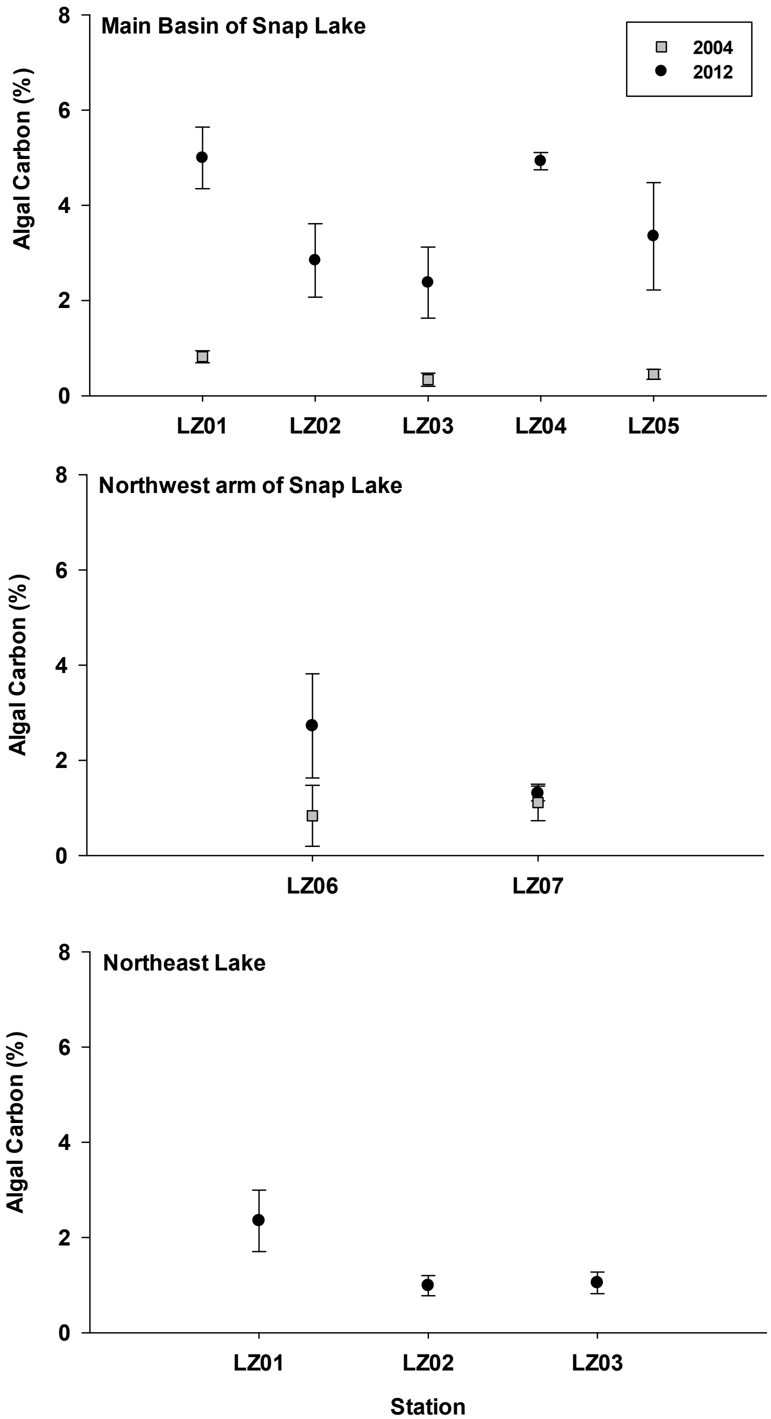
Figure 12.1-12 Spatial Trends in the Ratio of Carbon to Chlorophyll *a*, Epilithic Algae, Snap Lake 2004 and 2012, Northeast Lake 2012



Note: Error bars represent standard error of the means; Boundary of nutrient deficiency based on Healey and Hendzel (1980).

C = carbon; Chl *a* = chlorophyll *a*.

Figure 12.1-13 Percentage of Epilithic Algal Carbon, Snap Lake 2004 and 2012, Northeast Lake 2012



Note: Error bars represent standard error of the means.
% = percent.

12.1.5.5 Epilithic Algal Community Composition

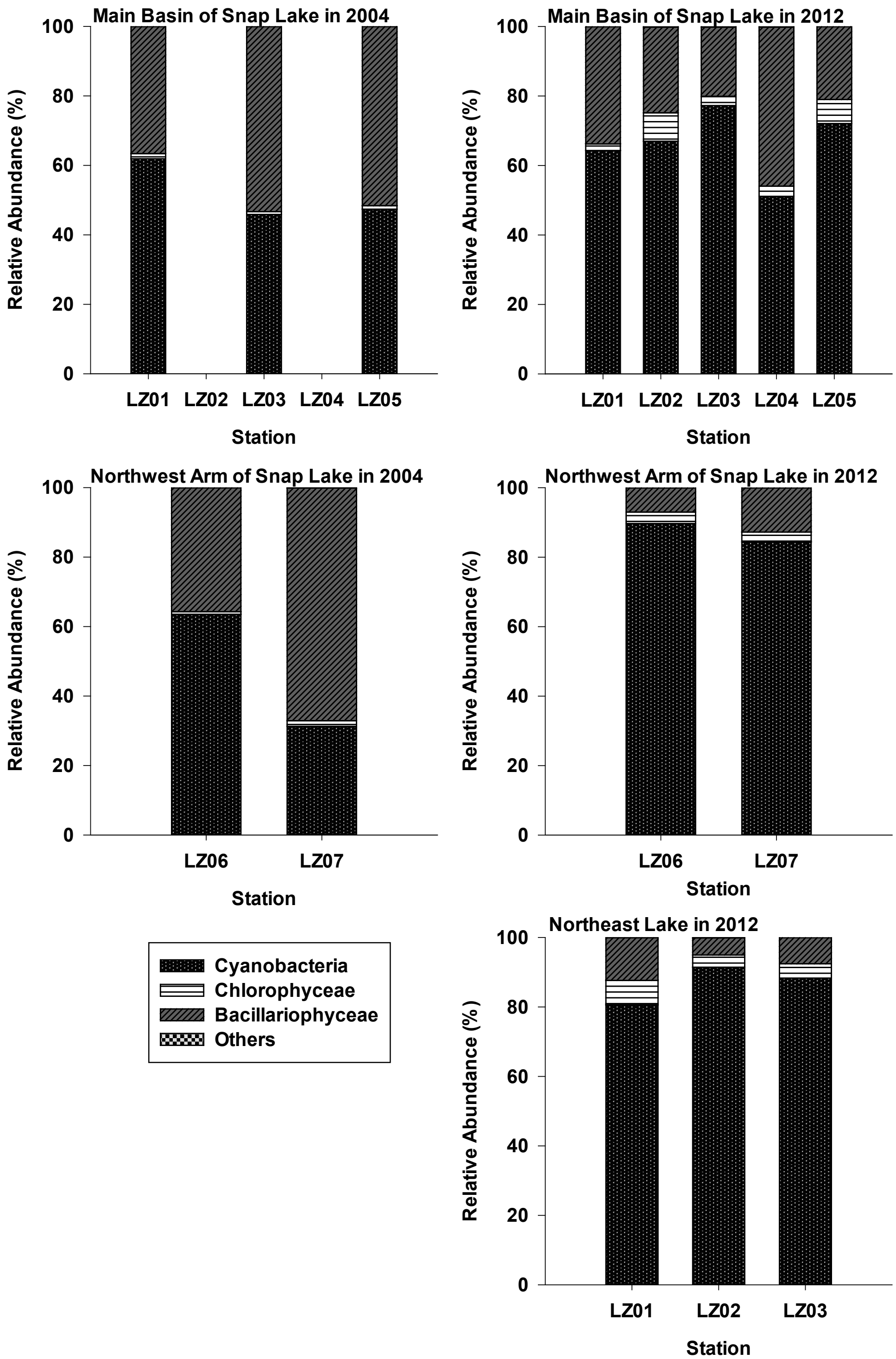
In 2004, epilithic algal abundance was dominated by either cyanobacteria or Bacillariophyceae (diatoms) at all stations in Snap Lake (Figure 12.1-14). In 2012, the epilithic algae in both Snap Lake and Northeast Lake were cyanobacteria-dominated. With the exception of SNAP LZ01 (PERI 04 in 2004) in the main basin of Snap Lake, cyanobacteria relative abundance was greater in 2012 compared to 2004. The relative abundance of chlorophytes was greater in 2004 compared to 2012, with similar abundances found at most stations in Snap Lake and Northeast Lake.

In 2012, relative abundances of the major epilithic algal groups were similar in Northeast Lake and the northwest arm of Snap Lake (Figure 12.1-14). Relative diatom abundance was greater in the main basin of Snap Lake compared to the northwest arm of Snap Lake and Northeast Lake.

Relative epilithic algal biomass was dominated by cyanobacteria at all stations in 2004, except SNAP LZ07 in the northwest arm of Snap Lake, which was dominated by diatoms (Figure 12.1-15). Diatoms and chlorophytes were present in 2004, but at low relative abundances compared to cyanobacteria.

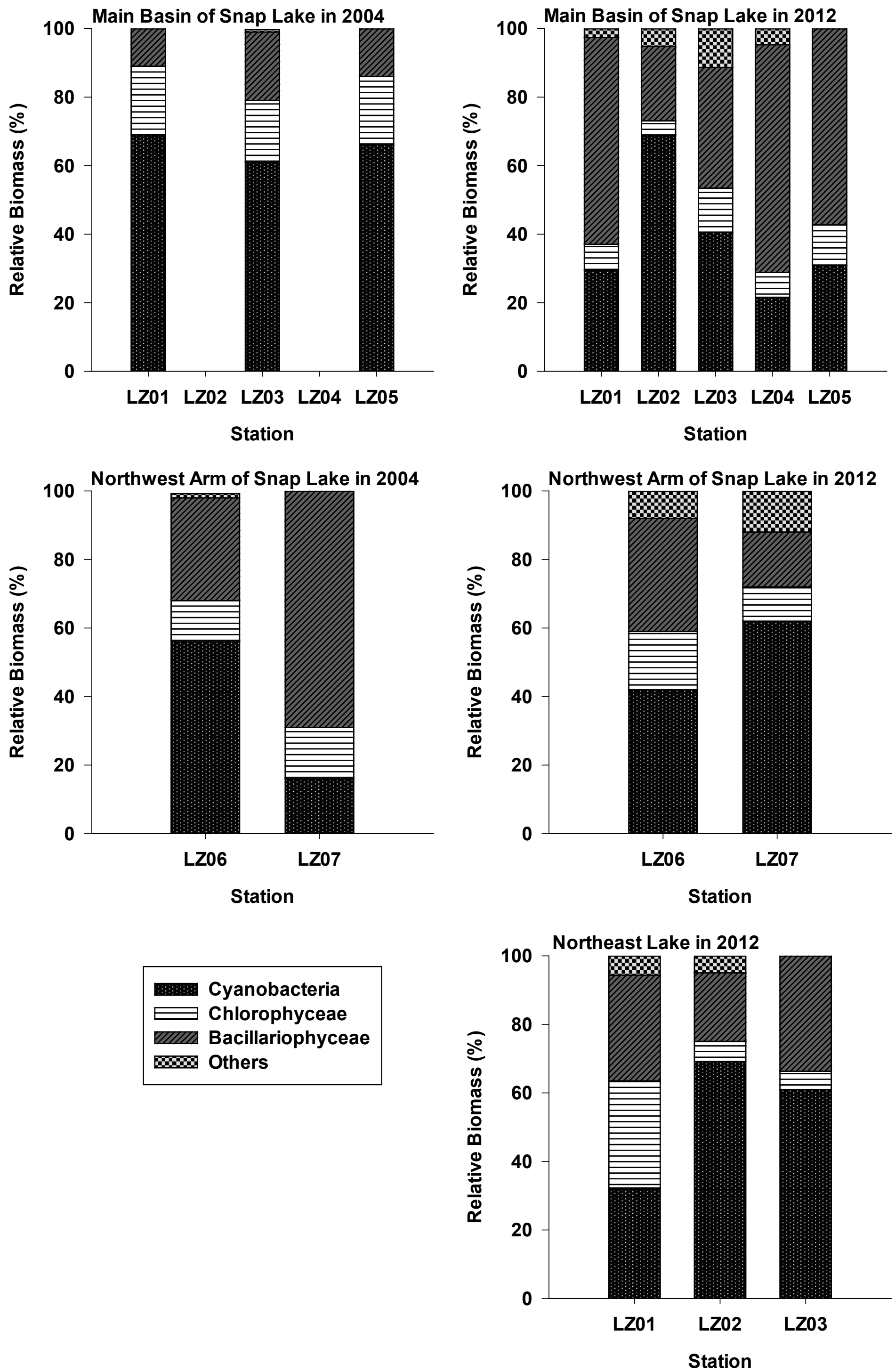
In 2012, relative biomass of the major epilithic algal groups differed among the main basin of Snap Lake, the northwest arm of Snap Lake, and Northeast Lake, and also varied among stations within each sampling area. Three stations in the main basin of Snap Lake (SNAP LZ01, SNAP LZ04, and SNAP LZ05) were diatom-dominated with secondary dominance by cyanobacteria. Two stations in Snap Lake, SNAP LZ03 in the main basin and SNAP LZ06 in the northwest arm, had lower proportions of diatoms and higher overall proportions of the other groups. Stations SNAP LZ02 in the main basin and SNAP LZ07 in northwest arm had higher overall biomass of cyanobacteria, with relatively low proportions of diatoms, chlorophytes, and "others." Relative percent composition at these two stations was similar to NEL LZ02 in Northeast Lake. NEL LZ03 was also similar to SNAP LZ02, SNAP LZ07, and NEL LZ02, except for a lack of "others" found at that station. NEL LZ01 differed markedly from the other Northeast Lake stations in having nearly equal proportions of chlorophytes, diatoms, and cyanobacteria.

Figure 12.1-14 Relative Abundance of Major Epilithic Algal Groups, Snap Lake 2004 and 2012, Northeast Lake 2012



Note: Sampling did not occur in Northeast Lake in 2004.
% = percent.

Figure 12.1-15 Relative Biomass of Major Epilithic Algal Groups, Snap Lake 2004 and 2012, Northeast Lake 2012



Note: Sampling did not occur in Northeast Lake in 2004.
% = percent.

12.1.5.6 Littoral Zone Invertebrate Community Composition

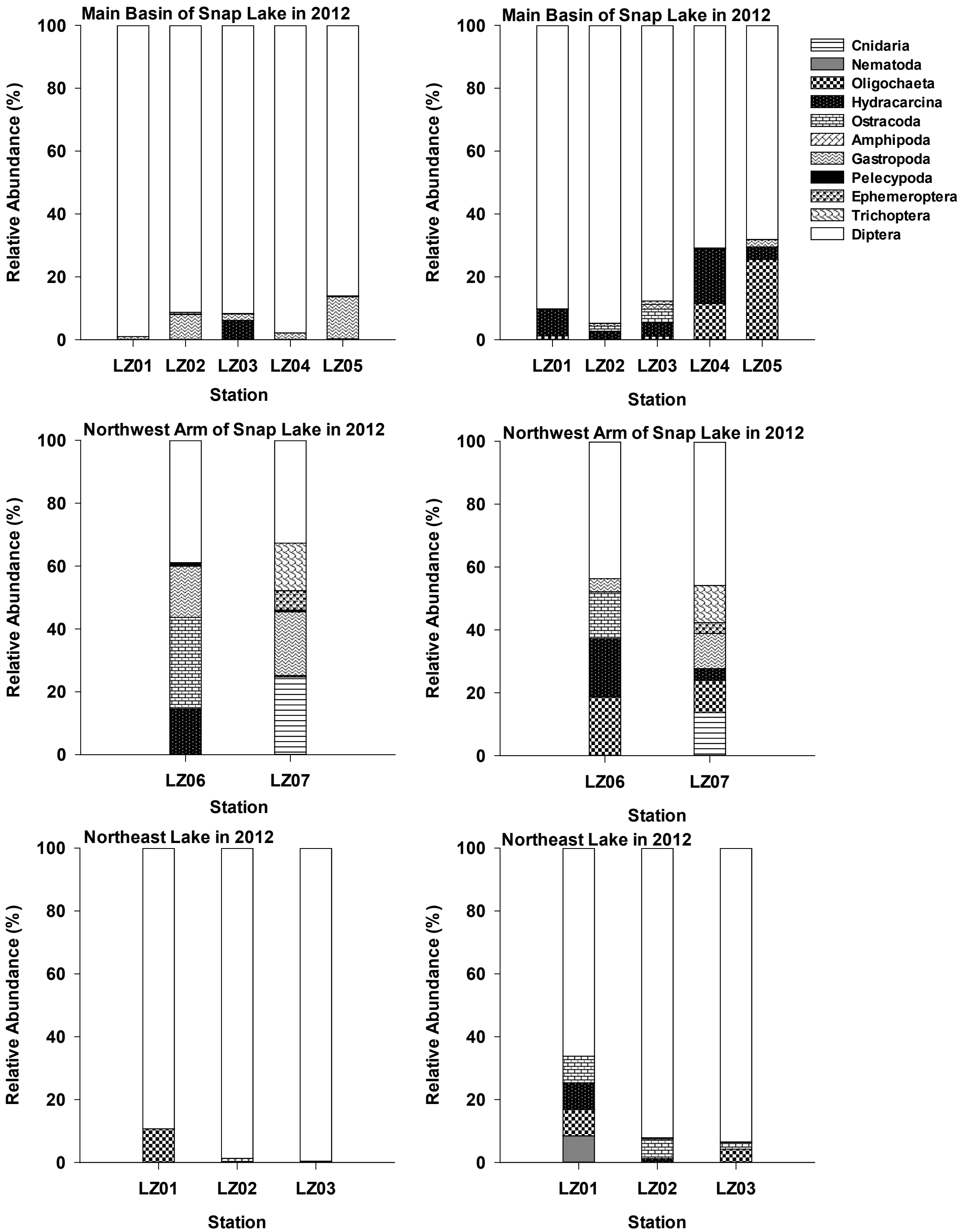
Mesh Size Assessment

The 250 µm mesh sieve captured a greater number and variety of invertebrates compared to the 500 µm mesh sieve in both lakes (Figure 12.1-16 and Appendix 12A.3 Table 12A.3-4). Composition of the invertebrate community differed based on mesh size, with slightly lower dominance of Chironomidae and greater diversity in the 250 µm mesh data set. Total abundance of captured invertebrates and total taxa were 5.6 and 1.2 times higher in the 250 µm mesh data set compared to the 500 µm mesh data, respectively.

Based on pooled data across all stations, several taxa (Nematoda, Naidinae, immature Orthocladiinae, and the midge genera *Stempellinella*, *Corynoneura* and *Monodiamesa*) were not present in the 500 µm mesh data set, indicating these invertebrates are too small to be captured by this mesh. Of these taxa, Naidinae was an abundant taxon in the 250 µm mesh data set, accounting for approximately 5% of total invertebrates. Abundances of Enchytraeidae, and the midge genera *Ablabesmyia*, *Dicrotendipes*, *Cladotanytarsus*, and *Cricotopus/Orthocladius* were two to five times higher in the 250 µm mesh data set, while Hydracarina, Ostracoda, immature Tanypodinae, and the midge genera *Parasmittia*, *Psectrocladius*, and *Tanytarsus* were six to 17 times more abundant in the 250 µm mesh data set. Other taxa were either present in equal numbers, or the difference between the two data sets was less than a factor of two.

Results of the mesh size evaluation indicate that a 250 µm mesh sieve should be used during future littoral zone sampling and sample processing, because this mesh retains a substantially larger number of invertebrates and taxa, thereby providing a more accurate representation of the littoral invertebrate community. The community assessment presented below was based on the 250 µm mesh data (i.e., pooled data for invertebrates retained in both 250 and 500 µm sieves).

Figure 12.1-16 Relative Abundances of Major Littoral Zone Invertebrate Groups in the Main Basin and Northwest Arm of Snap Lake and Northeast Lake, 2012



Note:% = percent.

Community Assessment

The common invertebrate taxa observed in the littoral zones of Snap Lake and Northeast Lake in 2012 were Cnidaria (hydras), Nematoda (roundworms), Oligochaeta (aquatic worms), Hydracarina (aquatic mites), Ostracoda (ostracods), Amphipoda (amphipods), Gastropoda (snails), Pelecypoda (fingernail clams), Ephemeroptera (mayflies), Trichoptera (caddisflies), and Diptera (true flies, mostly Chironomidae, or midges). At the lowest taxonomic level of identification, 32 taxa were identified in Snap Lake and 16 taxa in Northeast Lake (Appendix 12A.3 Table 12A.3-4). The lower number of taxa in Northeast Lake likely reflects the lower number of stations sampled in this lake. The most common genera in both lakes were the midges *Ablabesmyia* sp., *Thienemannimyia* sp., *Dicrotendipes* sp., *Cladotanytarsus* sp., *Paratanytarsus* sp., *Tanytarsus* sp., and *Psectrocladius* sp.

Diptera (mostly midges) dominated the invertebrate communities in the both Snap Lake and in Northeast Lake, while communities in the northwest arm of Snap Lake were more balanced (Figure 12.1-16). Higher relative densities of Ephemeroptera were observed at SNAP LZ04 and SNAP LZ05 in the main basin and SNAP LZ06 in the northwest arm compared to other stations. *Hydra* sp. (Cnidaria) were identified at only two stations in Snap Lake, while nematodes were present at only one station in Northeast Lake.

The majority of the Chironomidae density in the littoral zone in the main basin of Snap Lake consisted of the Tanytarsini tribe (Figure 12.1-17). Higher relative densities of Tanypodinae and Orthocladiinae subfamilies were observed in the littoral zones in Northeast Lake and the northwest arm of Snap Lake compared to the main basin of Snap Lake.

The distribution of littoral invertebrates in functional feeding groups indicated that the community in the main basin of Snap Lake was dominated by collector-gatherers, scrapers, filterers, and predators (Figure 12.1-18). The community in the northwest arm of Snap Lake was dominated by collector-gatherers and predators, while the community in Northeast Lake appeared to be evenly distributed among collector-gatherers, scrapers, filterers, and predators. Few grazers were present in the samples; however, the divers noted snail presence at a number of stations in both lakes. As previously noted (Section 12.1.1.1), increased calcium concentrations in water can result in increased snail populations in littoral zones of lakes; sampling in 2013 will attempt to quantitatively examine differences in snail densities between the lakes.

Based on the samples collected in 2012, there was no obvious difference in littoral invertebrate community composition or functional feeding group composition between the main basin of Snap Lake and Northwest Lake, with the possible exception of greater relative abundances of predators and herbivores in Northeast Lake. Community composition in terms of functional feeding groups was similar at all stations in the main basin of Snap Lake, suggesting similar food type and availability to invertebrates throughout the lake perimeter.

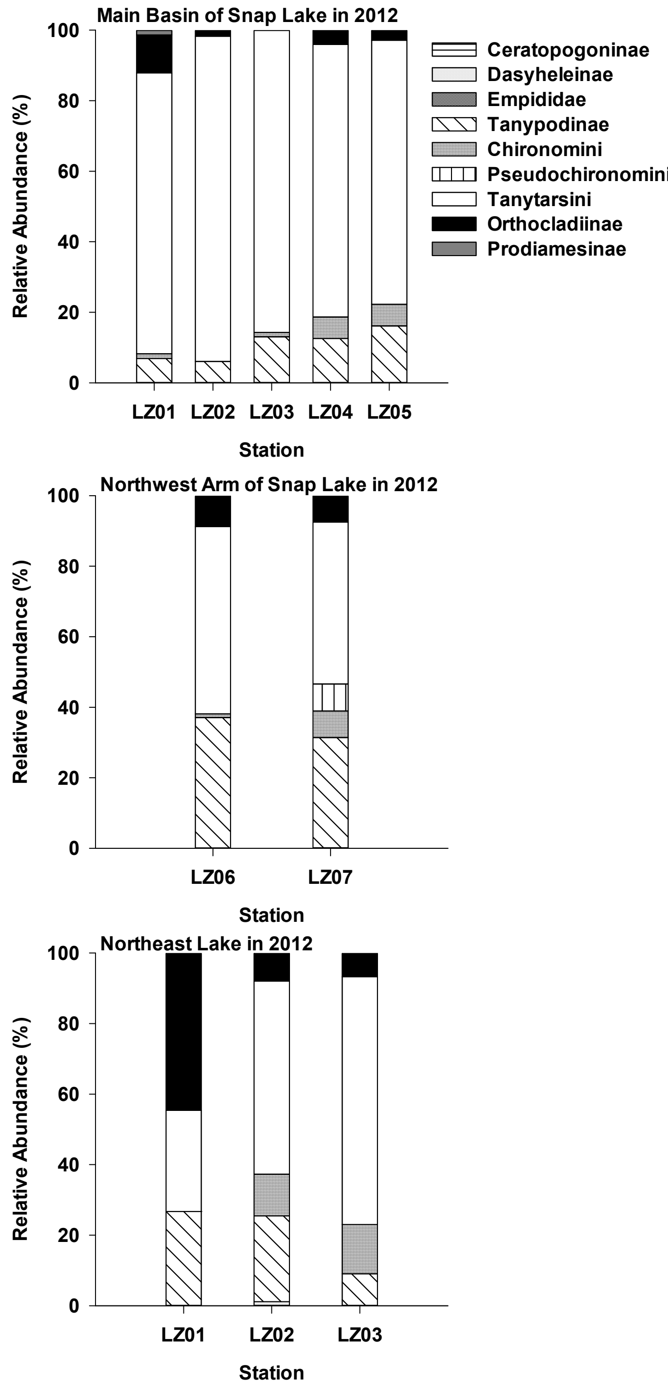
These results are subject to uncertainty related to the potential selectivity of the sampling method, which likely resulted in under-representation of attached or shelled taxa (e.g., snails and caddisflies). Snails were observed on rock surfaces in the littoral zone and some caddisflies were present in the samples collected using the sweep net technique. To collect quantitative information on the presence and abundance of these and other invertebrates, consideration of artificial substrata is recommended for future littoral zone invertebrate sampling, potentially in combination with the sweep net method used in 2012. Artificial substrata retrieved after a suitable colonization period would allow quantitative sampling of littoral zone invertebrates, including attached invertebrates. Sweep net sampling in combination with artificial substrata would allow documenting the full diversity of the littoral zone invertebrate community, including small, mobile invertebrates such as snails.

12.1.5.7 Within-Station and Among-Lake Variation in the Epilithic Algae and Associated Bacteria and Detritus

Within-station variation was high at most stations in 2012 (Appendix 12A.3, Table 12A.3-5). Out of the 60 samples investigated, using a variable-station combination of six variables (i.e., particulate N, P, C, chlorophyll *a*, epilithic algal biomass and abundance), 35 samples showed within-station variation greater than 20%. The mean CV for all stations combined was 32%. There were also lake-specific differences; the main basin of Snap Lake showed the highest variability (39%) followed by the northwest arm (32%) and Northeast Lake (21%), which may be a reflection of the higher number of sampling stations in the main basin.

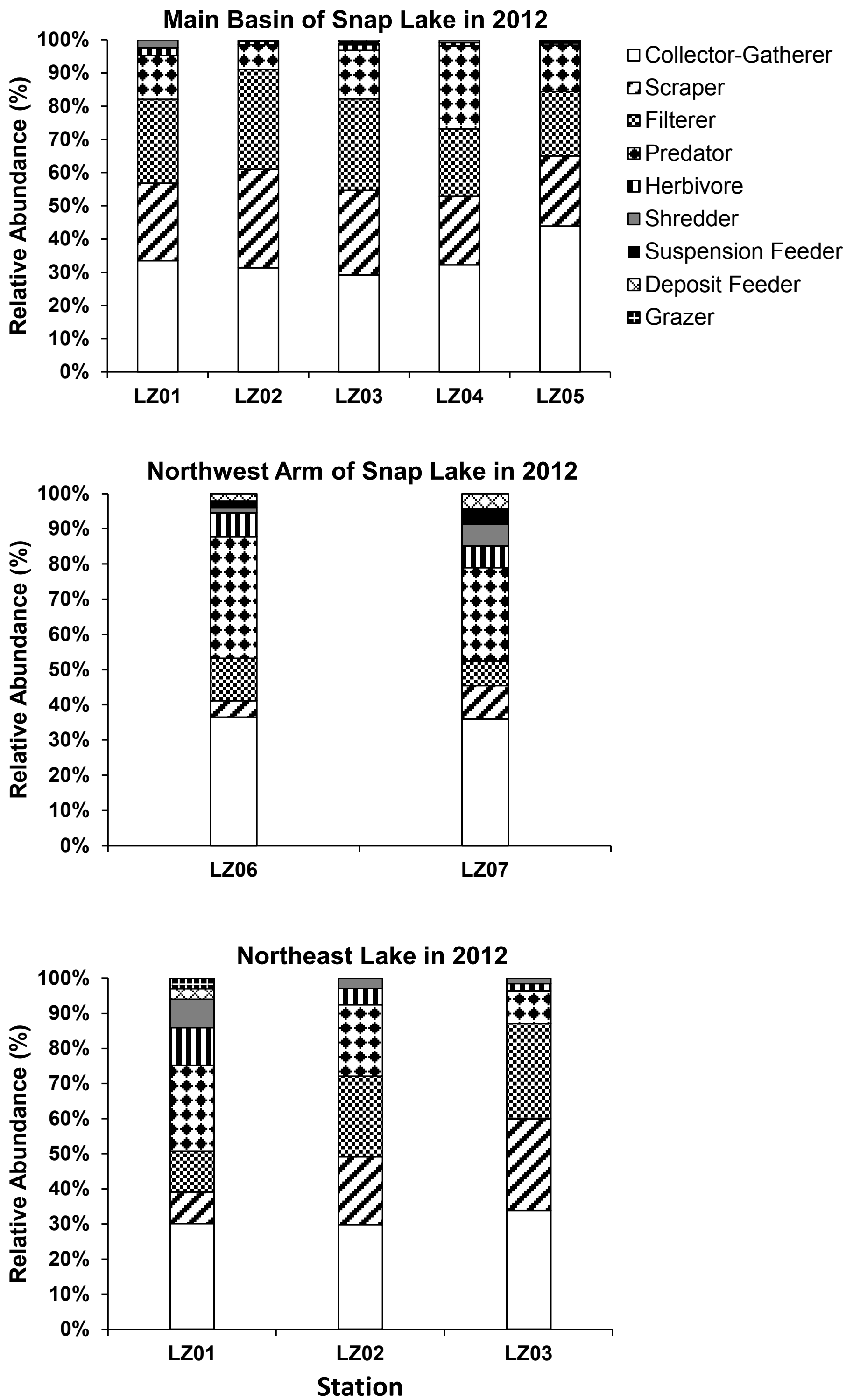
The greatest variation among stations based on all measured parameters was observed at SNAP LZ03 in the main basin of Snap Lake, with CVs ranging from 33% to 88%, followed by SNAP LZ02 in the main basin, where CVs ranged from 10% to 66% (Appendix 12A.3, Table 12A.3-5). Other stations showed high variability in one or more parameters, but not all parameters.

Figure 12.1-17 Relative Abundances of Diptera Taxa in the Main Basin and Northwest Arm of Snap Lake and Northeast Lake, 2012



Note:% = percent.

Figure 12.1-18 Relative Abundance of Invertebrate Functional Feeding Groups in the Main Basin and Northwest Arm of Snap Lake and Northeast Lake, 2012



Note:% = percent.

12.1.6 Discussion

12.1.6.1 Littoral Zone Nutrients

The DIC concentrations observed in Snap Lake and Northeast Lake in 2012 were higher than those generally observed in unaltered Pre-Cambrian Shield lakes at the ELA (less than 200 µmol/L; Hille 2008). The concentrations in Snap Lake and Northeast Lake were similar to those measured during an aquaculture-eutrophication experiment at the ELA (300 to 400 µmol/L; Hille 2008). Concentrations of DIC in the overlying water during that experiment were high enough that the system was not limited by DIC and, therefore, the supply of DIC for photosynthesis was not restricted by the boundary layer, such that P-limitation did not adversely affect epilithic algae (Turner et al. 1994). This may also be the case in Snap Lake and Northeast Lake, as littoral zone nutrient ratios indicate nutrient deficiency, particularly in P. The C:N, C:P, N:P, and C:chlorophyll *a* molar ratios in Snap Lake and Northeast Lake indicate nutrient deficiency, and the C:P and N:P nutrient molar ratios indicate that the limiting nutrient is P.

The molar ratios of C:N and C:P indicated higher nutrient deficiency, especially in P, and poorer food quality in the epilithic algal communities in Northeast Lake compared to Snap Lake. The mean C:P molar ratio at SNAP LZ03 in the main basin indicated both a decrease in P-deficiency and an increase in food quality. In contrast, the mean C:P molar ratio increased at SNAP LZ07 in the northwest arm of Snap Lake, which indicated an increase in P-deficiency and a decrease in food quality. The discrepancy observed between these two stations from 2004 to 2012 may be an indication of Mine-related P additions to the Snap Lake main basin, as opposed to the northwest arm. In addition, the C:N molar ratio was lower in 2012 at all stations in Snap Lake compared to 2004, indicating a change towards greater nutrient sufficiency in 2012. This may also be an indication of Mine-related nutrient enrichment in the main basin of Snap Lake.

In high DIC lakes, where P remains limiting, an increase in P-loading can initially increase epilithic algal biomass and productivity (Fairchild and Lowe 1984; Cattaneo 1987), depending on the form and mechanism of P delivery (Cattaneo 1987; Wetzel 2001). Given that the epilithic algae in Snap Lake were severely P-limited before Mine start-up, and that nitrogen concentrations have subsequently increased, a slight increase in P-loading to the system is expected to have little effect on the epilithic algae. This is because the increased nitrogen concentrations are likely to continue to pull the community towards P-limitation. However, the decrease in nutrient deficiency, particularly N-deficiency, observed in 2012 compared to 2004 may be an indication of a Mine-related nutrient enrichment effect in Snap Lake.

12.1.6.2 Epilithic Algae and Associated Bacteria and Detritus

The settled volume of particulate matter at each station can be used as a measure of the amount of material, both biotic (living and dead) and abiotic. Higher settled volumes were observed in Northeast Lake compared to Snap Lake. In addition, higher settled volumes were observed in the northwest arm of Snap Lake, compared to the main basin. Settled volumes were not measured in 2004.

The percentages of algal C observed in Snap Lake in 2004 and in Northeast Lake in 2012 are similar to values observed in undisturbed lakes at the ELA (i.e., 1 to 5%; Hille 2008). Values similar to those observed in the main basin of Snap Lake in 2012 (i.e., values greater than 5%) were observed in a system receiving P-loading at the ELA (Hille 2008). The higher proportion of viable algal C in 2012 is an indication of higher food quality to grazers in the main basin of Snap Lake compared to 2004, and higher food quality in Snap Lake compared to Northeast Lake. In addition, the higher percent algal C observed in the main basin of Snap Lake in 2012 compared to 2004 may also be an indication of a Mine-related nutrient enrichment effect in Snap Lake.

Epilithic algal community composition differed between 2004 and 2012 in Snap Lake. In addition, a different algal community was observed in Snap Lake compared to Northeast Lake. Station-to-station variability in relative percent composition was greater in 2012 compared to 2004. In 2004, relative abundance and biomass were dominated by either cyanobacteria or diatoms at all stations in Snap Lake. In 2012, stations in the main basin of Snap Lake were generally diatom-dominated with secondary dominance by cyanobacteria, while stations in the northwest arm of Snap Lake and Northeast Lake were dominated by cyanobacteria.

With the observed increase in N-loading and increasing N:P ratio in Snap Lake, a shift in community composition away from cyanobacteria, and towards diatoms and chlorophytes is expected (Wetzel 2001), because the N-fixing cyanobacteria no longer have a competitive advantage over the other algal groups (Wehr and Sheath 2003). Accordingly, between 2004 and 2012, relative percent cyanobacteria biomass decreased in the main basin of Snap Lake, while the relative percent biomass of “others” and diatoms increased. A similar trend was also observed in the phytoplankton community (Section 5). The increase in diatom biomass is likely associated with both an increase in the N:P ratio and increased Si concentrations in the lake water, related to the treated effluent discharged from the Mine (Section 5).

The decrease in cyanobacterial biomass in the main basin in Snap Lake can also be linked to the observed increases in food quality (i.e., the decrease in the C:P ratio and increases in the percent algal C observed in the main basin). Cyanobacteria are generally considered a poor food source to grazers; because they produce toxins, their cells are often protected by mucilaginous sheaths and they form large inedible filaments (Haney 1987). Small unicellular chlorophytes and diatoms, on the other hand, are considered to be a better food source and higher food quality to littoral grazers (De Beers 2012b).

12.1.6.3 Littoral Zone Invertebrates

Results of the mesh size evaluation indicate that a 250 µm mesh sieve should be used during future littoral sampling and sample processing, because this mesh retains a substantially larger number of invertebrates and taxa, thereby providing a more accurate representation of the littoral invertebrate community.

Using the results of the 250 μm mesh sieve, it was found that Diptera (mostly Chironomidae) dominated the littoral zone invertebrate communities in Snap Lake and Northeast Lake. This is similar to the composition of the deep-water benthic invertebrate community, as documented by the 2012 AEMP results (Section 6), which also reported Chironomidae dominance. However, it may in part reflect selectivity of the sampling method towards small, light, and easily dislodged invertebrates (Section 12.1.6.6). Chironomidae accounted for 45% to 93% of the total density at deep water stations in 2011 (De Beers 2012b). Dominance of benthic communities by the Chironomidae is expected in the sub-Arctic region, where Snap Lake is located (Beaty et al. 2006; Northington et al. 2010).

The majority of the Chironomidae density in the littoral zone in the main basin of Snap Lake consisted of the Tanytarsini tribe, whereas the deep-water benthic community had high relative densities of both the Chironomini and Tanytarsini tribes (De Beers 2012b). Higher relative densities of Tanypodinae and Orthocladiinae subfamilies were observed in the littoral zones in Northeast Lake and the northwest arm of Snap Lake, compared to the main basin of Snap Lake.

Littoral invertebrate grazers can exert a strong top-down influence on the epilithic algae (Vadeboncoeur et al. 2002). Up to 50% of the diet of littoral invertebrates can be epilithic algae (Strayer and Likens 1986). Littoral grazers can cause community structure changes through the preferential removal or avoidance of certain algal species and changes in primary productivity and biomass (Smith et al. 2001). Understanding the extent to which littoral grazers use and incorporate epilithic algal biomass will help in the interpretation of how the Mine may be affecting the epilithic algal community. The Stable Isotope Special Study, planned for 2013, is expected to provide information needed to understand the extent to which epilithic algae are being used by higher trophic levels.

12.1.6.4 Within Station and Among Lake Variation in the Epilithic Algae and Associated Bacteria and Detritus

It is important to understand the degree of station-to-station variation when interpreting changes in epilithic algal biomass and composition. A low coefficient of variation, both within each area (i.e., main basin of Snap Lake, northwest arm of Snap Lake, and Northeast Lake) and within each station, is optimal. High coefficients of variation in nutrient concentrations and biomass may translate to inaccurate interpretation of effects, and prevent meaningful statistical analyses from being performed.

In 2012, five stations were sampled in the main basin of Snap Lake, two stations in the northwest arm of Snap Lake, and three stations in Northeast Lake. Within each of these stations, three sub-areas were sampled to examine within-station variation. Overall, differences between the main basin and northwest arm of Snap Lake and Northeast Lake were difficult to determine because of the small number of stations sampled. Larger sample sizes in the northwest arm of Snap Lake and Northeast Lake are recommended for the 2013 program.

Within-station variation was high at most stations in 2012. The high within-station variability in 2012 may have been partly caused by issues related to the sampling technique. Improved sampling techniques will be used in 2013.

Natural epilithic algae, along with their associated bacteria and detritus, such as those sampled in Snap Lake and Northeast Lake, are inherently variable and are a combination of extremely diverse microhabitats, which vary in developmental stages (Robinson 1983), as well as in composition among stations. In the early stages of community development, colonizing organisms spread randomly from the initially established clump of algae, creating variability among stations (Robinson 1983). In the late stages of development, variance increases with community density and thickness (Robinson 1983; Wetzel 2001). Variation can also be caused by changes in the influences of the non-algal components (e.g., bacteria and littoral grazers) at each station (Robinson 1983). Intense grazing can reduce chlorophyll concentrations, reduce algal cell densities, and alter community composition (Frost et al. 2002). In addition, differences in light availability, and patchy distribution of light through the euphotic zone can cause variation in primary productivity among stations and, thus, variability in biomass and energy flow (Wetzel 2001).

Understanding variation caused by light differences among stations is essential to understanding epilithic algal variability, as it is expected that light fields will vary within Snap Lake because of the treated effluent plume (De Beers 2012c). Therefore, it is recommended that light measurements and examinations of the attenuation coefficient be performed at each littoral zone sampling station in 2013.

12.1.6.5 Sampling Methods and QA/QC

Since sampling methods were suboptimal in 2012, it is important to rectify this during the 2013 sampling program. Ideally, future littoral zone sampling scheduling should include a day prior to sampling for diver training, including an in-water exercise with video documentation. At least one scientifically-trained diver should be involved in future sampling.

Further stabilization of variance is desirable, this can be accomplished by increasing the number of sub-areas sampled from three to five, or by sampling additional sub-areas at a subset of stations. Analytical variance should also be estimated to understand within and among-lake variability; this could be done in 2013 by sampling in duplicate at some stations.

The epilithic algal samples were prepared using lake water collected from the surface at each of the lakes. This water is used to bring the sample suspension to the appropriate volume necessary for filtration (Section 12.1.2.3). It is possible for the lake water to contain plankton, which would confound the epilithic algal taxonomic results. Studies at the ELA have shown that plankton contamination in these samples is minimal (Hille 2008); however, it is important to confirm that there are no plankton in the littoral zone samples collected from Snap Lake. A rapid microscope assessment of the lake water samples from both Snap Lake and Northeast Lake should be done in future to assess whether there are plankton present.

12.1.7 Conclusions

12.1.7.1 Key Question 1: Can littoral monitoring be conducted in Snap Lake and Northeast Lake, and does the inherent variability in the littoral zone allow the detection of Mine-related changes?

Epilithic algae have been widely used for monitoring in streams and rivers in temperate environments, whereas their use as an indicator of lake health in the Arctic is new and requires investigation. The present study showed that littoral zone monitoring can indeed be conducted in Snap and Northeast Lakes. Differences were apparent in the epilithic algal community and its associated bacteria and detritus between 2004 and 2012, and between Snap Lake and Northeast Lake. Continuation of this Special Study for the further two years is required to adequately answer the question as to whether Mine-related changes can be detected.

12.1.7.2 Key Question 2: What are the current ratios of particulate C:N, C:P, N:P, and C: chlorophyll *a,* and what is the current percent algal carbon in the littoral zones of the main basin of Snap Lake, northwest arm of Snap Lake, and Northeast Lake? How do these values compare to baseline and what do these values indicate about Mine-related changes in nutrient status and food quality for invertebrates and fish?

The C:N, C:P, N:P, and C:chlorophyll *a* molar ratios in Snap Lake and Northeast Lake indicate nutrient deficiency. The C:P and N:P nutrient molar ratios point towards P as the limiting nutrient, as also observed in lake water AEMP monitoring. The molar ratios of C:N and C:P indicated higher nutrient deficiency, especially in P, and poorer food quality in the epilithic algal communities in Northeast Lake compared to Snap Lake. The C:N molar ratio decreased between 2004 and 2012 at all stations in Snap Lake, indicating a change towards greater nutrient sufficiency in 2012. This could indicate Mine-related nutrient enrichment of the epilithic algae and associated bacteria in the main basin of Snap Lake. If so, this may also indicate additional food for invertebrates and fish.

The percentage of viable algal carbon has increased in the main basin of Snap Lake since 2004, but remained similar between 2004 and 2012 in the northwest arm of Snap Lake. The increase in the proportion of viable algal carbon in Snap Lake is an indication of increased food quality to grazers in the main basin of Snap Lake. The higher percent algal carbon values and, therefore, improved food quality, observed in the littoral zone of the main basin of Snap Lake in 2012 may also be an indication of Mine-related nutrient enrichment in the main basin of Snap Lake. However, continuation of this Special Study for the further two years is required before the question regarding significance of Mine-related changes on food quality for invertebrates and fish can be adequately answered.

12.1.7.3 Key Question 3: What is the current status, in terms of relative abundance and relative biomass, of the epilithic algal communities in the main basin of Snap Lake, northwest arm of Snap Lake, and Northeast Lake? Do these results provide any evidence of a Mine-related effect?

Epilithic algal community composition differed between 2004 and 2012 in Snap Lake. In addition, a different algal community was observed in Snap Lake compared to Northeast Lake. In 2004, relative abundance and biomass were dominated by either cyanobacteria or diatoms at all stations in Snap Lake. In 2012, stations in the main basin of Snap Lake were generally diatom-dominated with secondary dominance by cyanobacteria, while stations in the northwest arm of Snap Lake and Northeast Lake were dominated by cyanobacteria. However, continuation of this Special Study for the further two years is required before the question of a Mine-related effect and its significance can be adequately answered.

12.1.7.4 Key Question 4: What is the current invertebrate composition in the littoral zones of the main basin of Snap Lake, northwest arm of Snap Lake, and Northeast Lake? Do these results provide any evidence of a Mine-related effect?

Diptera dominated the invertebrate communities in the main basin and northwest arm of Snap Lake and Northeast Lake. Diptera densities were greater in Northeast Lake compared to Snap Lake. Within the Diptera, the Chironomidae family dominated in Snap Lake and Northeast Lake. Most of the Chironomidae density in the littoral zone in the main basin of Snap Lake consisted of the Tanytarsini tribe, whereas in the deep-water benthic community high densities of both Chironomini and Tanytarsini tribes were observed (De Beers 2012b). Higher densities of Tanypodinae and Orthocladinae subfamilies were observed in the littoral zones in Northeast Lake and the northwest arm of Snap Lake compared to main basin of Snap Lake. However, continuation of this Special Study for the further two years is required before the question of a Mine-related effect and its significance can be adequately answered.

12.1.8 Recommendations

The 2012 Littoral Zone Special Study findings showed that littoral zone sampling can be conducted in Snap Lake and may provide useful data. However, based on experience from the 2012 sampling, refinements to the study design are recommended to improve the program and better characterize within-station variability:

- Additional training of divers is recommended prior to sampling: in-water training and evaluation; at least one of the divers should be scientifically-trained; and, resampling if initial sampling is not conducted properly.
- Duplicate samples should be collected to allow for an estimate of analytical variance.
- A rapid microscope assessment should be conducted of the lake water samples from both Snap Lake and Northeast Lake to assess the presence of plankton.
- One more sampling station should be added in the northwest arm of Snap Lake to increase spatial coverage through the gradient of treated effluent exposure present in this part of the lake. Two more sampling stations should be added in Northeast Lake to increase statistical power and spatial coverage.
- A more detailed taxonomic evaluation is recommended, i.e., providing the taxonomist used in 2004 a subset of samples from 2013 to identify and enumerate.
- Light in the water column should be measured using underwater light meters along with examination of the attenuation coefficient at each littoral sampling station.
- A 250 µm mesh sieve should be used for collection of littoral zone biota.
- Artificial substrates should be deployed for invertebrate sampling, in combination with sweep net sampling. Using artificial substrates would allow quantitative sampling of littoral zone invertebrates, including attached taxa, while sweep net sampling would allow documenting the full diversity of the littoral zone invertebrate community.

12.1.9 References

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12.2 DOWNSTREAM LAKES SPECIAL STUDY

12.2.1 Introduction

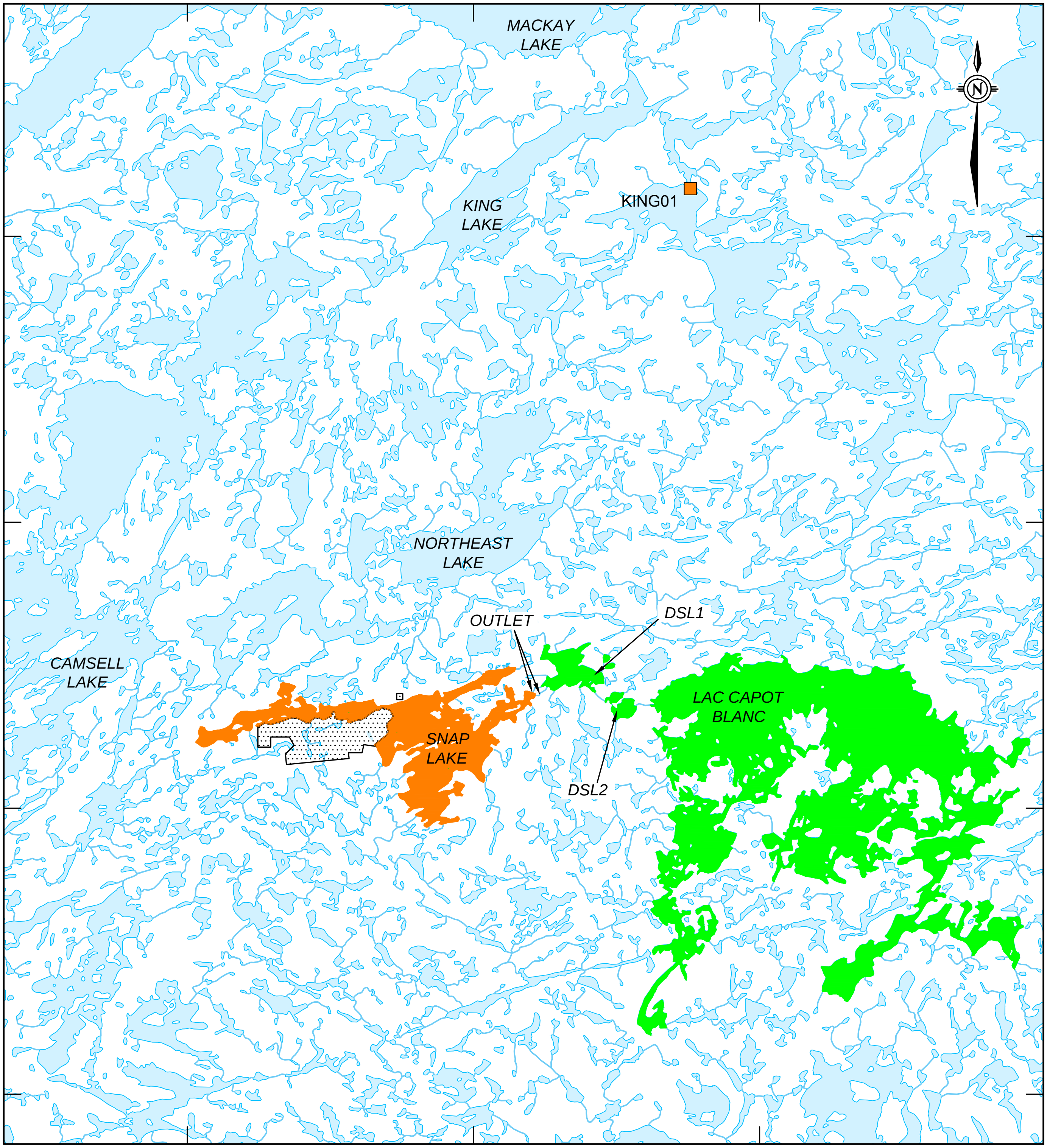
12.2.1.1 Background

De Beers Canada Inc. (De Beers) owns and operates the Snap Lake Mine (Mine), a diamond mine located approximately 220 kilometres (km) northeast of Yellowknife, Northwest Territories. As part of the Mine's operations, treated effluent is discharged into Snap Lake, which is located 30 km south of MacKay Lake and west of Lac Capot Blanc (Figure 12.2-1).

The Mine operates under a Class A Water Licence (#MV2011L2 0004) issued by the Mackenzie Valley Land and Water Board (MVLWB 2012), and the Aquatic Effects Monitoring Program (AEMP) fulfills a requirement under Part G of the Water Licence. The AEMP is designed to monitor Snap Lake for Mine related effects, to verify and update the Environmental Assessment Report (EAR) predictions (De Beers 2002), and to support and inform management decisions made by the Mine.

The AEMP includes one downstream monitoring station at King Lake (KING01), approximately 25 km downstream of Snap Lake (Figure 12.2-1). Between 2004 and 2011, there was no evidence of treated effluent at King Lake (De Beers 2012a). The extent of treated effluent in the lakes immediately downstream of Snap Lake was not well known however. In preparation for updating the AEMP Design Plan, initial reconnaissance work was completed downstream of Snap Lake in 2011 to determine whether, as predicted, treated effluent was present in lakes immediately downstream of Snap Lake. Until 2011, the spatial extent of the treated effluent downstream of Snap Lake had not been delineated.

Results of the initial 2011 downstream reconnaissance sampling program indicated that concentrations of total dissolved solids (TDS), and by extension, other Mine-related constituents (i.e., major ions and nitrate) decreased with distance downstream from Snap Lake, consistent with the EAR and modelling predictions (De Beers 2002; Golder 2011). In 2011, evidence of the treated effluent was detected throughout downstream lakes 1 and 2, and near the inlet of Lac Capot Blanc. Field conductivity at the inlet of Lac Capot Blanc was 188 microSiemens per centimetre (µS/cm), and declined to approximately 30 µS/cm within an approximately 50 metres (m) distance from the inlet. Concentrations of TDS and nitrate were at background levels within 330 m from the inlet. Concentrations of Mine-related constituents reached background within 6 km downstream of Snap Lake. In the EAR, concentrations were conservatively predicted to reach near background concentrations approximately 44 km downstream of Snap Lake at the end of operations, using a steady-state mixing model and assuming maximum concentrations during operations.



LEGEND

- AEMP WATER QUALITY STATION
- SNAP LAKE MINE FOOTPRINT
- MONITORING AREA
- TARGETED DOWNSTREAM MONITORING 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.

4 0 4

SCALE KILOMETRES

PROJECT

DE BEERS GROUP OF COMPANIES

TITLE

DOWNSTREAM LAKES OVERVIEW, 2012

Golder Associates

PROJECT	13-1349-0001.6200	
DESIGN	TD	2013-03-13
CADD	JEF	2013-03-30
CHECK	LMB	2013-04-19
REVIEW	PMC	2013-04-19

FILE No13134900016200A12.2-1

SCALE AS SHOWN REV. 0

FIGURE 12.2-1

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Downstream monitoring was continued in 2012 as a Special Study to the AEMP to further document the extent of the treated effluent downstream of Snap Lake, and to begin preliminary collection of bathymetric, water quality, sediment, plankton, and benthic invertebrate data in the first three lakes downstream of Snap Lake (Figure 12.2-1) in preparation for identifying future AEMP study stations. The intent of this section of the 2012 AEMP Report is to summarize the results from the 2012 Downstream Lakes Special Study and provide recommendations for future investigations downstream of Snap Lake.

12.2.1.2 Objectives

The objectives of the 2012 Downstream Lakes Special Study were:

- to collect additional baseline data (i.e., water and sediment quality, plankton and benthic invertebrates) in the three lakes downstream of Snap Lake;
- to estimate the spatial extent of the treated effluent plume downstream of Snap Lake; and,
- to update bathymetry information collected in 2011.

This information will aid in the selection of future AEMP study stations and will be used to validate the existing downstream lake water quality model.

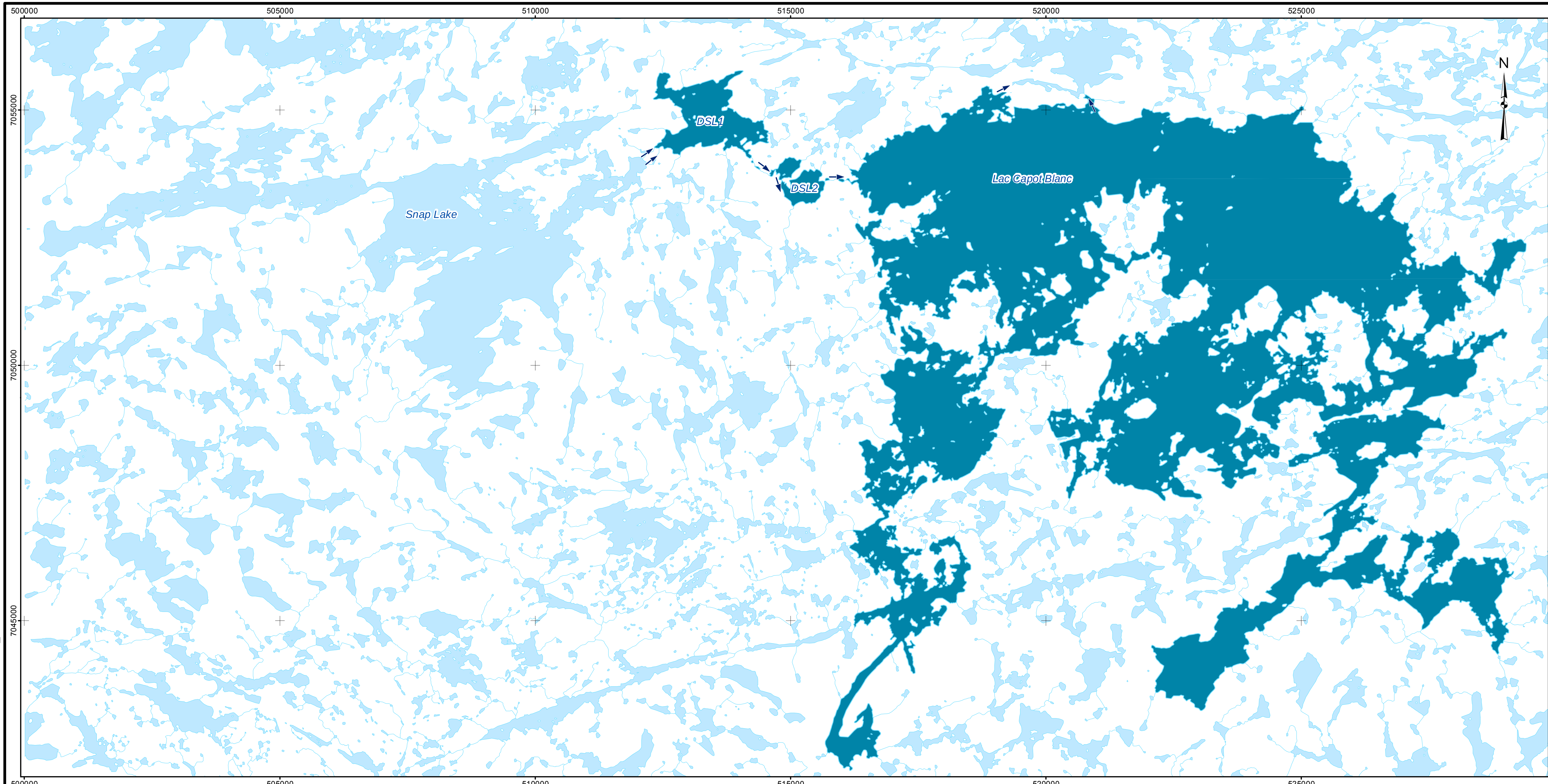
Based on these objectives, this Special Study addressed three key questions:

1. What is the spatial extent of the treated effluent plume downstream of Snap Lake?
2. What are the current water and sediment quality characteristics in the three downstream lakes?
3. What is the current benthic invertebrate community composition in the three downstream lakes?

Plankton samples were collected in 2012 and archived. The intention was to have representative plankton samples should future comparisons be required. A final study design including plankton AEMP study stations will be developed once the special study is complete.

12.2.2 Study Area

Three lakes located immediately downstream of Snap Lake were surveyed during the 2012 downstream sampling program, based on evidence of treated effluent in these lakes during the 2011 downstream reconnaissance work (De Beers 2012a). For the purpose of this Special Study, the lakes are referred to as downstream lakes 1 and 2 (abbreviated as DSL1 and DSL2, respectively), and Lac Capot Blanc. Outflow from Snap Lake passes through two flume structures at the lake outlet (i.e., Hydrology Station 1 [H1] and Hydrology Station 2 [H2]; Appendix 12B.1, Photos 12B.1-1 and 12B.12) and two small ponds before reaching DSL1. The main flow path is then to DSL2, Lac Capot Blanc, and downstream through the Lockhart River watershed (Figure 12.2-2).



LEGEND

- FLOW DIRECTION
- WATERCOURSE
- LAKES OF INTEREST
- WATERBODY

3 0 3

SCALE 1:75,000 KILOMETRES

NOTE

DSL1: FIRST LAKE DOWNSTREAM OF SNAP LAKE.
DSL2: SECOND LAKE DOWNSTREAM OF SNAP LAKE.

REFERENCE

BASE VECTOR DIGITAL CANVEC 1:50K DATA OBTAINED FROM GEOGRATIS ONLINE. ©1992 NATURAL RESOURCES CANADA, CENTRE FOR TOPOGRAPHIC INFORMATION. ALL RIGHTS RESERVED. DATUM: NAD83 PROJECTION: UTM ZONE 12

PROJECT

DE BEERS
GROUP OF COMPANIES

TITLE

2012 DOWNSTREAM LAKES STUDY AREA

Golder Associates

PROJECT		12-1337-0002	FILE No.	
DESIGN	DR	03 Oct. 2012	SCALE AS SHOWN	REV. 0
GIS	CH	01 Mar. 2013		
CHECK	LMB	19 Apr. 2013	FIGURE: 12.2-2	
REVIEW	PMC	18 Apr. 2013		

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12.2.3 Methods

The field component of this Special Study was conducted in August 2012, and included field water quality measurements at the inlet and outlet of each lake (pH, conductivity, dissolved oxygen [DO], and water temperature), collection of water quality samples from selected inlet and outlet locations, and spatial delineation of the treated effluent plume using conductivity (Table 12.2-1). Three stations (one deep location per lake) were sampled for water quality, plankton, benthic invertebrates, and sediment quality. An update to the 2011 bathymetric survey was completed during the 2012 field program in DSL2 and the northwest basin of Lac Capot Blanc.

12.2.3.1 Bathymetry

No additional bathymetry transect data were required in DSL1, because sufficient coverage was achieved during the 2011 survey (Appendix 12B.2; Figures 12B.2-1 to 12B.2-3). Bathymetry transects were completed in DSL2 on August 22 and in Lac Capot Blanc on August 23 to 28, 2012, in a grid fashion using a sonar coupled with a global positioning system (GPS) unit (sonar/GPS).

A full bathymetric survey was completed in DSL2, with transects spaced between 40 to 50 m apart. Lac Capot Blanc required additional transects to update the existing map, with a focus on the northwest basin where the inlet stream is located, and continuing to the 2nd outlet (Appendix 12B.2; Figure 12B.2-3). Transects were spaced approximately 50 to 100 m apart in the NW basin. Additional bathymetric transects, using coarser grid spacing, were completed in the northeast and southeast basins. Spacing of these coarser transects varied from 250 to 500 m on average, with some spaced at 600 m apart.

Transect layout consisted of longitudinal transects along the long axis of each lake, crossed by lateral transects across the width of each lake. Longitudinal and lateral transects were approximately equally spaced across the width and length of each lake to provide as much detail as possible. Data were stored in the boat-mounted sonar/GPS and downloaded each day onto a computer as a Garmin MapSource file.

Table 12.2-1 Downstream Lake Sampling Locations, August 2012

Lake	Station	Easting[(a)]	Northing[(a)]	Depth (m)	Field Measurement Type	Sample Type
DSL1	Inlet DSL1	512308	7054241	0.3	Surface	Surface Water Sample
	DSL1-B3	512564	7054384	2.3	Water Column Profile	-
	DSL1-1	513403	7054940	14.5	Water Column Profile	Mid-depth Water Sample, Plankton, Benthic Invertebrates, Sediment
	DSL1-E3	513936	7054339	1.9	Water Column Profile	-
	Outlet DSL1	514211	7054083	0.3	Surface Measurement	-
DSL2	Inlet DSL2	514816	7053620	0.3	Surface Measurement	Surface Water Sample
	DSL2-1	515197	7053418	6.8	Water Column Profile	Mid-depth Water Sample, Plankton, Benthic Invertebrates, Sediment
	Outlet DSL2	515832	7053620	0.3	Surface	-
Lac Capot Blanc	Inlet 1 LCB	516297	7053619	0.3	Surface	Surface Water Sample
	Inlet 2 LCB	515912	7053655	0.8	Surface, Bottom	-
	LCB-A1	516316	7053748	1.9	Water Column Profile	-
	LCB-A2	516332	7053685	0.7	Water Column Profile	-
	LCB-A3	516348	7053622	2	Water Column Profile	-
	LCB-A4	516363	7053560	3.1	Water Column Profile	-
	LCB-A5	516379	7053497	0.6	Water Column Profile	-
	LCB-A5A	516356	7053518	3.1	Water Column Profile	-
	LCB-A6	516449	7053465	3.6	Water Column Profile	-
	LCB-B1	516443	7053866	2.5	Surface, Bottom Measurements	-
	LCB-B3	516475	7053650	2.6	Water Column Profile	-
	LCB-B5	516537	7053493	6	Surface, Bottom	-
	LCB-B6	516589	7053490	5.2	Surface, Bottom	-
	LCB-C1	516715	7054019	7.2	Water Column Profile	-
	LCB-C3	516803	7053666	9	Water Column Profile	-
	LCB-C5	516890	7053313	3.6	Surface, Bottom	-
	LCB-C7	516757	7053465	4	Surface, Bottom	-
	LCB-D1	518162	7054414	7.5	Surface, Bottom	-
	LCB-1	518411	7053352	13.9	Water Column Profile	Mid-depth Water Sample, Plankton, Benthic invertebrates, Sediment
	LCB-2	523306	7053340	4.1	Water Column Profile	-
	LCB-3	525105	7052028	11	Water Column Profile	-
	LCB-4	522634	7049277	12	Water Column Profile	-
	Outlet 1 LCB	519152	7055364	0.3	Surface	Surface Water Sample
	Outlet 2 LCB	520623	7055390	0.3	Surface	Surface Water Sample
Unnamed Water-course	FF1	522941	7058616	0.2	Surface	-
	FF2	523249	7059786	0.3	Surface	-
	FF3	521181	7067043	0.2	Surface	-
	KING01	517583	7071651	0.2	Surface	-

Note: The letter in the station identifier denotes the lateral transect along which field measurements were collected; lateral transects were positioned across the width of the lake, perpendicular to the inlet-outlet flow path.

(a) Universal Transverse Mercator (UTM) Coordinates; North American Datum (NAD) 83; Zone 12V.

- = no samples collected; m = metre; DSL1 = first lake downstream of Snap Lake 1; DSL2 = second lake downstream of Snap Lake; LCB = Lac Capot Blanc.

12.2.3.2 Supporting Environmental Variables

During the field survey, the following supporting environmental information was recorded:

- sampling date and time;
- weather conditions, such as air temperature, wind velocity, and wind direction;
- the GPS coordinates recorded as Universal Transverse Mercator (UTM) for each station;
- water depth;
- Secchi depth;
- vertical profiles of water temperature, DO, pH, and conductivity, measured at discrete intervals (Section 3.3); and,
- water temperature data using three temperature loggers (Onset TidbiT Water Temperature Loggers – UTBI-001).

The temperature loggers were installed in all three downstream lakes starting July 10 and 11, 2012, and removed September 10 and 11, 2012. The temperature loggers were programed to record water temperature hourly.

One shallow site location (i.e., less than 1.0 m depth) and one deep site location (i.e., water depth of 10 to 15 m) were selected in each lake. At the shallow site, one temperature logger was installed mid-depth, approximately 0.5 m below the water surface. The deep sampling site had two temperature loggers installed on the same line and float; one logger was installed 0.3 m below the water surface, and the second logger was installed 1.0 m above the bottom substrate.

The shallow site logger for DSL1 was lost and no data were recovered during the retrieval period in September. The locations of the temperature loggers are shown in Figures 12.2-3 to 12.2-5, Section 12.2.3.2.

12.2.3.3 Water Quality Field Measurements

Field measurements of DO, pH, water temperature, and conductivity were collected using a YSI 650 Multiparameter Display System (MDS) water quality meter with a YSI 600 Quick Sample (QS) multi parameter water quality probe during the 2012 downstream sampling program. Water column profile data were collected at three stations in DSL1, one station in DSL2 and fourteen stations in Lac Capot Blanc (Table 12.2-1; Figures 12.2-3 to 12.2-5). A 30-m cable was connected to the YSI meter for depth profiles.

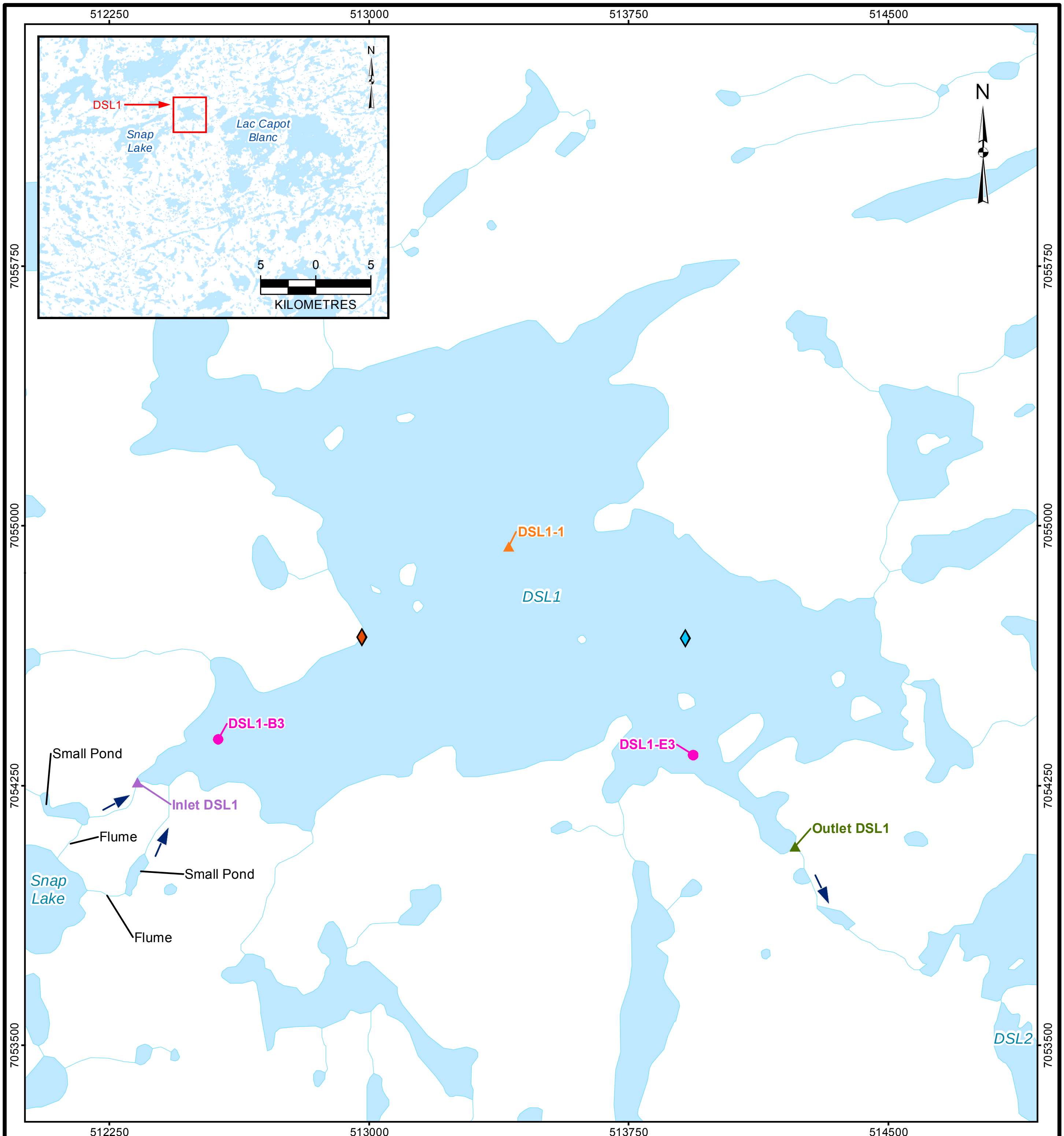
Field measurements were collected as follows:

- for shallow stations, spot field measurements at surface and/or bottom of the water column;
- for lake depths between 2.5 and 5.0 m, a measurement was recorded every 0.5 m within the water column; and,
- for lake depths greater than 5.0 m, a measurement was recorded every 1.0 m within the water column.

Secchi depth was measured using a 20 centimetre (cm) diameter circular plate known as a Secchi disk. The Secchi disk was lowered over the side of the boat, away from direct sunlight, to the depth at which it could no longer be seen. This depth was then recorded as the Secchi depth.

Station Naming Convention

The Downstream Lakes Special Study began in 2011 as a field reconnaissance program. A naming convention was assigned by the field crew based on transects set up in each of the lakes. In Table 12.2-1, the letters in the station identifiers refer the lateral transect along which field measurements were collected. At that time, the downstream lakes were referred to as Lake 1, Lake 2, and Lake 3. The 2011 naming convention was also used during the 2012 Downstream Lakes Special Study. Subsequent to the 2012 field program, the 2013 AEMP Design Plan was submitted, in which the downstream lakes naming convention was revised to reduce redundancy and improve consistency. Therefore, in this report and future programs, the revised naming convention is and will be used. A comparison of the station names is provided in Appendix 12B.3.



LEGEND

- PROFILE ONLY
- PROFILE + WATER SAMPLE, PLANKTON, BENTHIC INVERTEBRATES, SEDIMENT
- SURFACE FIELD MEASUREMENT ONLY
- SURFACE FIELD MEASUREMENT + WATER SAMPLE

TEMPERATURE LOGGER LOCATION

- DEEP
- SHALLOW

- FLOW DIRECTION
- WATERCOURSE
- WATERBODY

0.5 0 0.5

SCALE 1:16,000 KILOMETRES

NOTE

DSL1: FIRST LAKE DOWNSTREAM OF SNAP LAKE.
DSL2: SECOND LAKE DOWNSTREAM OF SNAP LAKE.

REFERENCE

BASE VECTOR DIGITAL CANVEC 1:50K DATA OBTAINED FROM GEOGRATIS ONLINE.
 DATUM: NAD83 PROJECTION: UTM ZONE 12

PROJECT

DE BEERS
GROUP OF COMPANIES

TITLE

2012 DOWNSTREAM SAMPLING PROGRAM, DOWNSTREAM LAKE 1 (DSL1)

Golder Associates

PROJECT		PROJECT NO.	FILE No.	
DESIGN	CW	5 Sep. 2012	SCALE AS SHOWN	REV. 2
GIS	CH	01 Mar, 2013		
CHECK	LMB	19 Apr. 2013	FIGURE: 12.2-3	
REVIEW	PMC	18 Apr. 2013		

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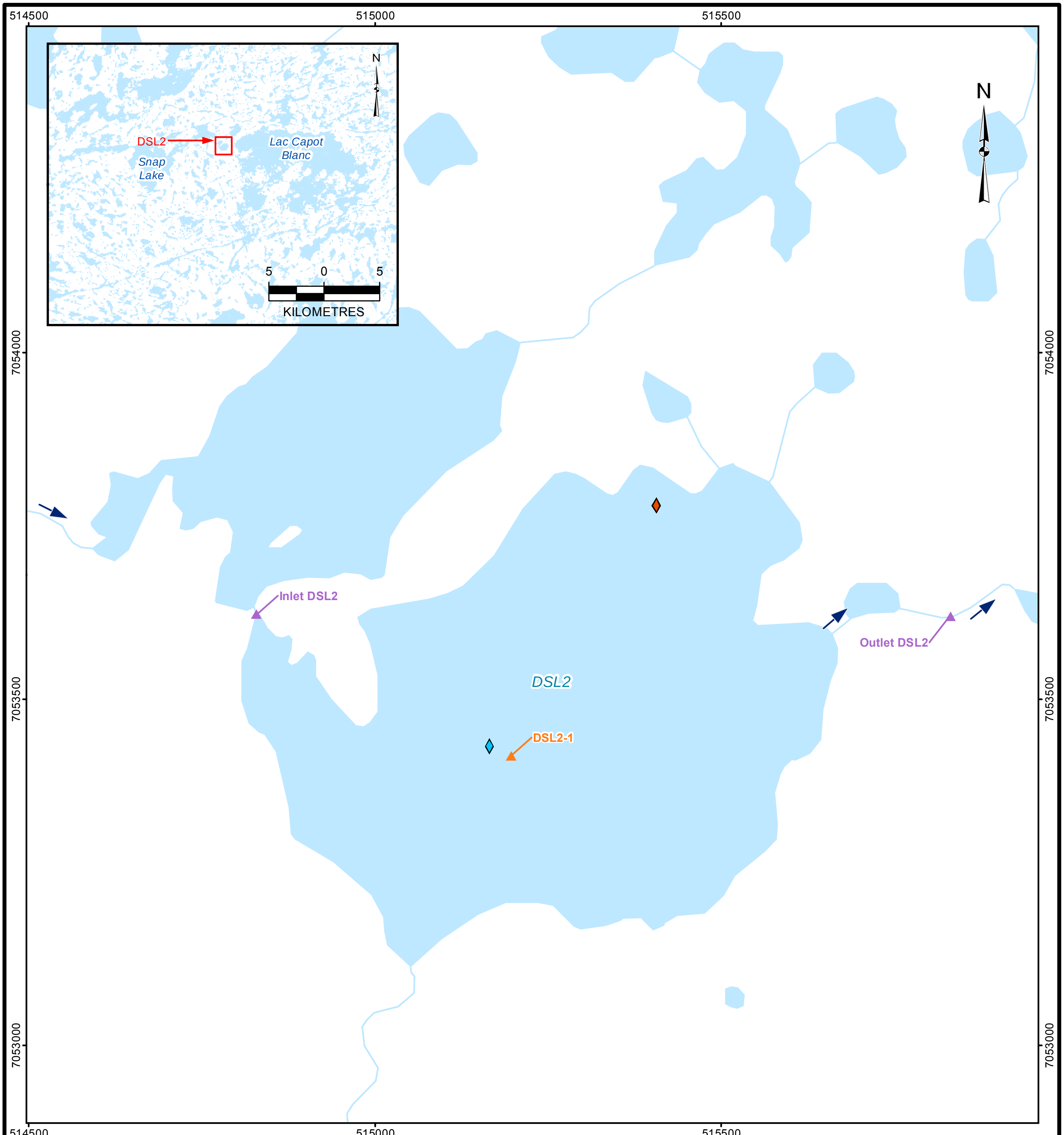

LEGEND

- PROFILE + WATER SAMPLE, PLANKTON, BENTHIC INVERTEBRATES, SEDIMENT
- SURFACE FIELD MEASUREMENT + WATER SAMPLE
- WATERBODY

TEMPERATURE LOGGER LOCATION

- DEEP
- SHALLOW
- FLOW DIRECTION
- WATERCOURSE

300 0 300

SCALE 1:8,000 METRES

NOTE

DSL2: SECOND LAKE DOWNSTREAM OF SNAP LAKE

REFERENCE

BASE VECTOR DIGITAL CANVEC 1:50K DATA OBTAINED FROM GEOGRATIS ONLINE.
 DATUM: NAD83 PROJECTION: UTM ZONE 12

PROJECT

DE BEERS
GROUP OF COMPANIES

TITLE

2012 DOWNSTREAM SAMPLING PROGRAM, DOWNSTREAM LAKE 2 (DSL2)

Golder Associates

PROJECT		PROJECT NO.	FILE No.	
DESIGN	CW	5 Sep.2012	SCALE AS SHOWN	REV. 2
GIS	CH	01 Mar. 2013	**FIGURE: 12.2-4**	
CHECK	LMB	19 Apr. 2013		
REVIEW	PMC	18 Apr. 2013		

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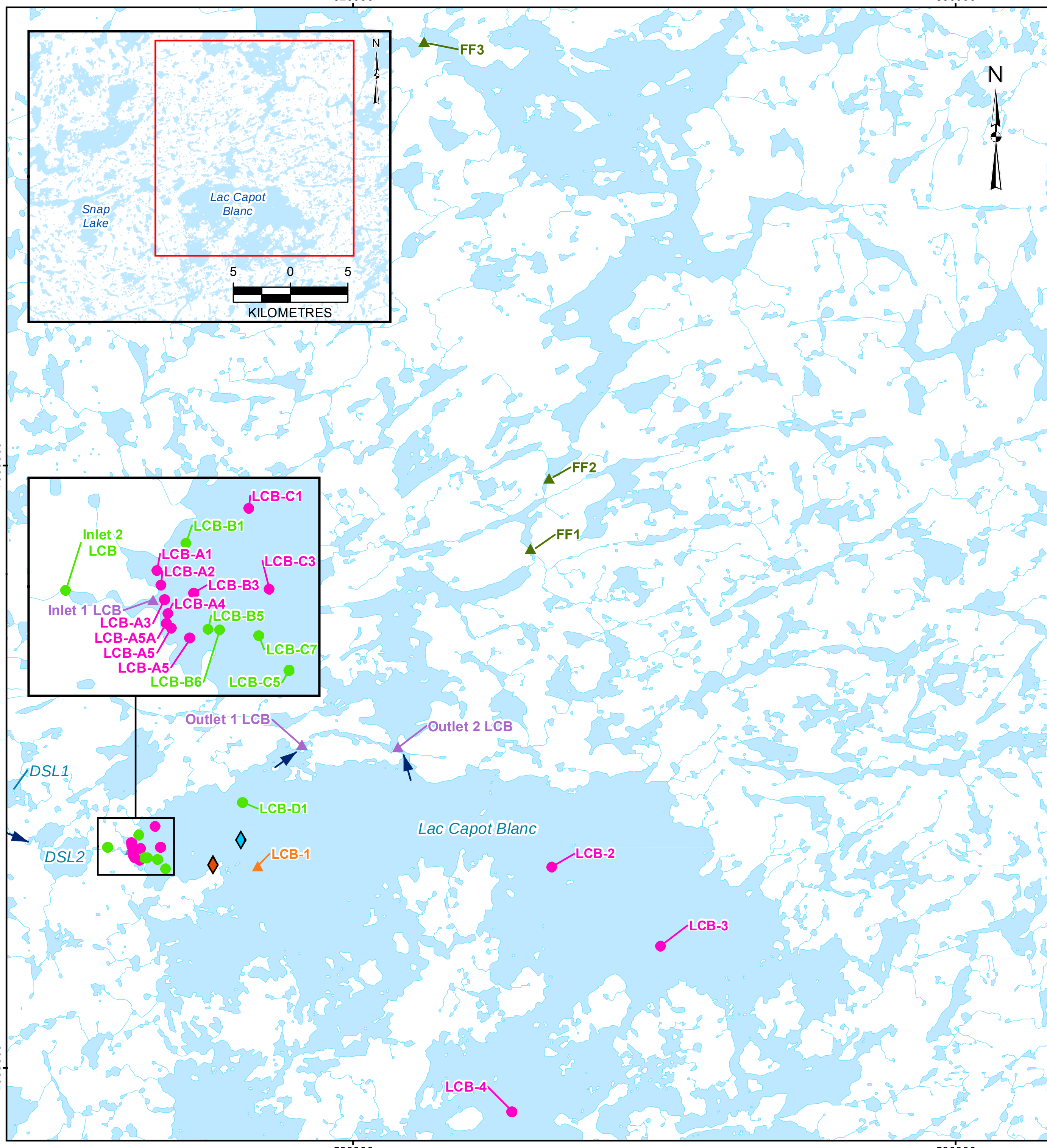

LEGEND

- PROFILE ONLY
- PROFILE + WATER SAMPLE, PLANKTON, BENTHIC INVERTEBRATES, SEDIMENT
- SURFACE + BOTTOM FIELD MEASUREMENT
- SURFACE FIELD MEASUREMENT ONLY
- SURFACE FIELD MEASUREMENT + WATER SAMPLE

TEMPERATURE LOGGER LOCATION

- DEEP
- SHALLOW

- FLOW DIRECTION
- WATERCOURSE
- WATERBODY

3 0 3

SCALE 1:95,000 KILOMETRES

NOTE

DSL1: FIRST LAKE DOWNSTREAM OF SNAP LAKE.
DSL2: SECOND LAKE DOWNSTREAM OF SNAP LAKE.

REFERENCE

BASE VECTOR DIGITAL CANVEC 1:50K DATA OBTAINED FROM GEOGRATIS ONLINE.
 DATUM: NAD83 PROJECTION: UTM ZONE 12

PROJECT

DE BEERS
GROUP OF COMPANIES

TITLE

2012 DOWNSTREAM SAMPLING PROGRAM, LAC CAPOT BLANC

Golder Associates

PROJECT		PROJECT NO.	FILE No.	
DESIGN	CW	5 Sep. 2012	SCALE AS SHOWN	REV. 4
GIS	JC	08 Mar. 2013		
CHECK	LMB	19 Apr. 2013	FIGURE: 12.2-5	
REVIEW	PMC	18 Apr. 2013		

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12.2.3.4 Water Sample Collection

Water samples were collected at eight stations, including one inlet and one deep water location in each lake, as well as the two outlet locations in Lac Capot Blanc (Table 12.2-1). At the inlet and outlet stations, grab water samples were collected from the middle of the watercourse, from 0.3 m below the surface. At the deep water locations, water samples were collected from mid-depth using a Teflon or PVC Kemmerer sampler, after taking profile measurements.

With the exception of sample bottles requiring filtration, bottles were filled directly in the field. At the inlet and outlet stations, bottles were filled directly, whereas at the deep water locations, bottles were filled from the Kemmerer sampler. Samples requiring filtration were collected in a 1 litre (L) biochemical oxygen demand bottle from either a Teflon and PVC Kemmerer for transport back to the De Beers environmental laboratory. Sample bottles were triple-rinsed with sample water before filling, with the exception of glass bottles. All bottles were then labelled with the sample station name, unique sample control number, sample depth and type of Kemmerer used. Preservatives were added to appropriate samples after filtering. Water samples were submitted to appropriate analytical laboratories and analyzed for conventional parameters, major ions, nutrients, and metals, following the AEMP parameter suite (De Beers 2012a).

Quality assurance (QA) and quality control (QC) protocols were followed as per the De Beers Snap Lake AEMP. The QC samples were:

- One travel blank, a set of pre-filled with laboratory-distilled de-ionized water (DDW) bottles provided by the laboratory. These bottles remained sealed and accompanied the field sample bottles at all times.
- One field blank, consisting of a set of sample bottles filled in the field with DDW.
- One set of duplicate samples, an additional set of sample bottles prepared from a second sample collected from the same location.

These samples were submitted to the appropriate analytical laboratories along with the field-collected samples.

12.2.3.5 Plankton Monitoring

Phytoplankton and zooplankton samples were collected in all three downstream lakes. The plankton survey was completed at one deep sampling location for each lake, corresponding with the water quality, benthic invertebrate, and sediment sampling locations (Table 12.2-1).

Duplicate zooplankton samples were collected at each station, for a total of six samples (i.e., two from each of DSL1, DSL2, and Lac Capot Blanc). Zooplankton samples were collected using a 12 inch diameter, 153 micrometre (µm) mesh plankton net using a vertical tow method, starting

1 m above the lake bottom. Zooplankton samples were then preserved with a half piece of an Alka-Seltzer tablet, followed by adding approximately 125 millilitre (mL) of 4 percent (%) buffered formalin to the 250 mL sample bottles.

Phytoplankton samples were collected from the euphotic zones in the downstream lakes which are generally the top 6 m of the water column, or less if the maximum water depth was less than 6 m. Water samples were collected using a Kemmerer water sampler from the surface and at 2 m intervals (i.e., surface, 2, 4, and 6 m). The three equal volumes of water were then combined in a bucket and mixed to create a homogeneous composite sample. A 250 mL sub-sample was collected in a pre-preserved amber Nalgene® bottle filled with 10 mL Dafano's and 2.5 mL Lugol's solution.

The remaining composite sample water was used to fill a 1-L amber Nalgene® bottle for chlorophyll *a* analysis. Two samples, using approximately 500 mL of water each, were filtered onto 47 millimetre (mm) GF/C filters using a glass filter tower and vacuum pump. Each filter was removed using forceps, folded in half, wrapped in aluminum foil, and frozen. The chlorophyll *a* samples were then shipped to the University of Alberta Biogeochemical Analytical Laboratory, in Edmonton, Alberta, where analyses were completed.

The phytoplankton and zooplankton samples collected as part of this Special Study were archived and will be analyzed for abundance, species composition, and biomass. The samples will be analyzed following further reconnaissance of the physical characteristics of the downstream lakes in 2014.

12.2.3.6 Sediment and Benthic Invertebrate Samples

Sampling Methods

Sediment and benthic invertebrate samples were collected at one station (Table 12.2-1) in each of the three downstream lakes. Sediment and benthic samples were collected from locations in the 10 to 15 m depth range from fine depositional sediments using an Ekman grab. The sediment and benthic invertebrate samples were collected after plankton sampling, water quality vertical profiles, and water quality sampling were completed. The following procedures, which are also used for routine AEMP sampling, were used to collect sediment and benthic invertebrate samples.

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 to allow for an estimate of within-station variability. Each sample was sieved and preserved separately for taxonomic analysis. Individual samples were placed in a 500 µm sieve bucket or bag. Each sample was flushed with lake water to remove the fine sediment, and the remaining material was transferred into a 500 mL pre-labelled bottle and preserved with 10% buffered formalin. Samples were shipped to J. Zloty, Ph.D. (Summerland, BC) for enumeration and taxonomic identification of invertebrates.

Three sediment samples were collected at each station, and the top 5 cm of surface sediment were removed from each grab and combined to generate a single composite sample. A field duplicate sample (composite of the top 5 cm of sediment from three additional grab samples) was collected at one randomly selected station (Station LCB-3 D3 in Lac Capot Blanc). Two 250 mL glass jars were filled from the composite sample for nutrients, carbon, and metals analyses. A large pre-labelled Ziploc bag with at least 500 mL of composite sample was also collected for particle size and moisture content analyses. Sediment samples were packed in a cooler with ice packs and shipped to the ALS Canada Ltd. (ALS, Edmonton, AB) analytical laboratory for analyses of the AEMP suite of sediment chemistry parameters.

Benthic Invertebrate Sample Sorting and Taxonomic Identification

Benthic invertebrate samples were processed according to standard protocols based on recommendations in Environment Canada (2002) and Gibbons et al. (1993). Benthic invertebrate samples were first washed through a 500 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 mm and 500 µm mesh size. Because 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; Wiederholm 1983; Oliver and Roussel 1983; Brinkhurst 1986; Pennak 1989; Clifford 1991; Coffman and Ferrington 1996; Merritt and Cummins 1996; Maschwitz and Cook 2000; Epler 2001). Organisms that could not be identified to the desired 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.

Benthic Invertebrate Data Summary

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 calanoid copepods (Calanoida), 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 stage of some Dipteran taxa are non-benthic. Abundance data received as number of organisms per sample were converted to density data consisting of number of organisms per square metre (org/m²). Unusual abundance data were validated before data summary and statistical analyses.

The following summary variables were calculated for each benthic invertebrate station:

- mean invertebrate density (org/m²);
- community composition as percentages of major taxa;
- Simpson's diversity index (diversity);
- taxonomic richness; and,
- evenness.

Summary statistics including the arithmetic mean and standard error (SE) were calculated for each of the above variables.

12.2.4 Results

12.2.4.1 Bathymetry

In DSL1, depth of near-shore waters ranged from 0 to 2 m, while depth in the majority of the lake ranged from 2 to 6 m. Two small areas of 14 to 16 m depth were located in the middle of DSL1 (Appendix 12B.2; Figure 12B.2-1). Bathymetry data collected from DSL2 indicated that the majority of this lake was less than 6 m deep (Appendix 12B.2, Figure 12B.2 2). One small area in the middle of the southern half of the lake was in the 6 to 8 m depth range. The maximum depth in Lac Capot Blanc was 36 m; however, the majority of the lake was less than 16 m deep (Appendix 12B.2; Figure 12B.2-3). In the northwest basin, where evidence of treated effluent was found, depths were less than 16 m. A few scattered deep areas in the main basin of Lac Capot Blanc reach 18 to 22 m; the deepest area in the lake is located in the middle of the northeast basin, at 34 to 36 m depth.

12.2.4.2 Water Quality

Quality Control Summary

The QC results from the 2012 Downstream Lakes Special Study indicated that:

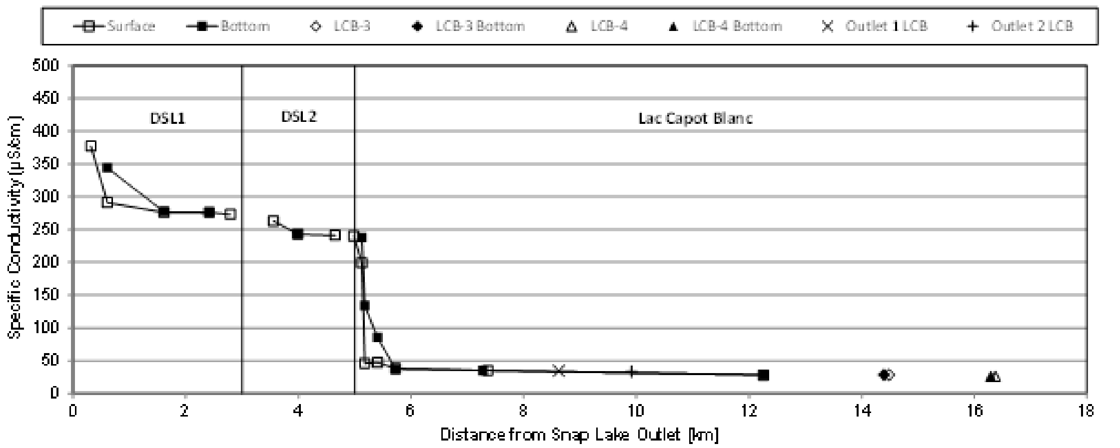
- The relative percent differences between the duplicate samples were within 20% for all parameters (Appendix 12B.4; Table 12B.4-1 and 12B.4-2). Therefore, the intra-site variability and field sampling precision were rated as low and high, respectively.
- Samples were generally free of contamination. Blank results were generally reported below the detection limit (DL), or less than 10% of the minimum lake concentrations. Exceptions were boron and zinc concentrations in the field blank, and boron and strontium concentrations in the travel blank. Boron in both blanks, and zinc concentrations in field blank were greater than 10% of the minimum lake concentrations. Therefore, potential boron and zinc contamination may have occurred, and data for those two parameters presented herein should be interpreted with this limitation in mind.
- Boron and zinc were qualified as “likely contaminated” (i.e., data flagged with NP) and “potentially contaminated” (i.e., data flagged with QP), respectively as part of the overall QA/QC assessment of the 2012 AEMP water quality data (Appendix 3A). Results for parameters with QP flags, such as total zinc, were less likely to be affected by potential contamination. Therefore, no further investigation into the potential zinc contamination was conducted. The source of boron contamination was, however investigated. Laboratory-DDW from Flett Research Limited, which was used for washing sampling equipment and filling field blanks for metals, was sent to ALS Canada Ltd. for analysis (Appendix 3A). Boron was detected in the DDW samples at concentrations near the levels detected in the 2012 blank samples collected as part of AEMP. The glass storage bottles were likely contributing to the detectable levels of boron in the blank samples (Appendix 3A). Therefore, a recommendation for the 2013 AEMP was to use plastic bottles to store the ALS DDW to reduce boron contamination in the blank samples (Appendix 3A).

Overall, the quality of water chemistry data collected during the 2012 downstream lakes program was acceptable and adequate to address the objectives of the study.

Spatial Delineation of Treated Effluent

Spatial delineation of treated effluent downstream of Snap Lake was assessed using field measurements in DSL1, DSL2, and Lac Capot Blanc during 2012. Emphasis was placed on conductivity, an indirect electrical measurement for Mine-related constituents including TDS, nitrate, and major ions. Field conductivity measurements were compared with those measured in other reference areas, such as the Northeast Lake, where conductivity has been consistently below 30 µS/cm. Stations located in Lac Capot Blanc, farthest from Snap Lake (i.e., LCB-3 and LCB-4), were lower than 30 µS/cm and considered background or reference values (Figure 12.2-6). Conductivity values above 30 µS/cm were, therefore, assumed to be influenced by treated effluent exposure.

Figure 12.2-6 Field Conductivity Downstream of Snap Lake, 2012



Note: DSL1 = first lake downstream of Snap Lake; DSL2 = second lake downstream of Snap Lake.
µS/cm = microSiemens per centimetre; km = kilometre.

Evidence of treated effluent was measured in DSL1, DSL2, and Lac Capot Blanc in 2012. Field conductivity measurements at stations Inlet DSL1, Inlet DSL2, and Inlet LCB were 377, 263, and 240 µS/cm, respectively (Figure 12.2-6). Conductivity notably decreased at LCB-C3, which is located approximately 650 m from the inlet of Lac Capot Blanc (Figure 12.2-5). At the farthest Lac Capot Blanc stations, LCB-3 and LCB-4, conductivity decreased to 28 and 26 µS/cm, respectively. Based on the 2012 conductivity values, the area influenced by treated effluent has increased since 2011 (Figure 12.2-7). In 2011, the field conductivity decreased to background levels within 50 m of the inlet (De Beers 2012a).

Results indicate that, in 2012, the extent of the treated effluent was limited to DSL1, DSL2, and Lac Capot Blanc. Field conductivity decreased from 240 µS/cm at the inlet of Lac Capot Blanc, to near background levels at the first and second outlets of Lac Capot Blanc (located 3.3 and 4.7 km northeast of the inlet, respectively [Figure 12.2-5]). Conductivity at Outlet 1 LCB and Outlet 2 LCB was 26 µS/cm. Four stations located downstream of Lac Capot Blanc were visited in 2012. Field conductivity at FF1, FF2, FF3, and KING01 ranged from 22 to 30 µS/cm. Field conductivity reached background within 6 km downstream of Snap Lake in 2012, as also observed in 2011 (Figure 12.2-6; Table 12.2-2). In the EAR (De Beers 2002), parameter concentrations associated with the treated effluent discharge were conservatively predicted to reach background concentrations within 44 km of Snap Lake by the end of operations, assuming maximum concentrations during operations.

Similar to field conductivity, concentrations of Mine-related constituents including TDS, nitrate, and major ions were higher in DSL1 and DSL2 compared to those measured at most stations of Lac Capot Blanc in 2012. Concentrations of TDS, nitrate, and major ions decreased notably at LCB-1 in Lac Capot Blanc. The same decreasing pattern was also observed in concentrations of total metals including barium, boron, molybdenum, rubidium, and strontium, which are also characteristic of the treated effluent.

Table 12.2-2 Change in Conductivity from 2011 to 2012

Lake	Station	Conductivity (µS/cm)		Percent Increase (%)
		2011	2012	
DSL1	Inlet DSL1	307	377	23
	Outlet DSL1	218	273	25
DSL2	Inlet DSL2	204	263	29
	Outlet DSL2	191	241	26
Lac Capot Blanc	Inlet 1 LCB	188	240	28
	Inlet 2 LCB	186	199	7
	Outlet 2 LCB	27	33	22

Source: De Beers (2012a).
µS/cm - microSiemens per centimeter; % = percent.

Vertical patterns were assessed using data collected at eighteen stations in DSL1, DSL2, and Lac Capot Blanc during 2012 (Figures 12.2-3 to 12.2-5 and Appendix 12B.5, Table 12B.5-1). There were generally no vertical trends in field conductivity, with some exceptions. Field conductivity measurements at DSL1-B3 and at the stations close to the inlet of Lac Capot Blanc (LCB-A3, LCB-A4, LCB-B5, and LCB-A5A) were elevated near the bottom of the water column (Table 12B.5-1). The results indicate that the denser treated effluent discharged from Snap Lake Mine tends to sink to the bottom of the water column as it enters DSL1 and Lac Capot Blanc, but then vertically mixes throughout the water column at other locations in those lakes. Vertical gradients were not observed in DSL2, a shallower lake, indicating vertical mixing in DSL2.

LEGEND

WATERCOURSE

EVIDENCE OF TREATED EFFLUENT (2011)

EVIDENCE OF TREATED EFFLUENT (2012)

WATERBODY

2 0 2

SCALE 1:50,000 KILOMETRES

NOTE

DSL1: FIRST LAKE DOWNSTREAM OF SNAP LAKE.

DSL2: SECOND LAKE DOWNSTREAM OF SNAP LAKE.

REFERENCE

BASE VECTOR DIGITAL CANVEC 1:50K DATA OBTAINED FROM GEOGRATIS ONLINE. ©1992 NATURAL RESOURCES CANADA, CENTRE FOR TOPOGRAPHIC INFORMATION. ALL RIGHTS RESERVED. DATUM: NAD83 PROJECTION: UTM ZONE 12

PROJECT

DE BEERS GROUP OF COMPANIES

TITLE

ZONE OF TREATED EFFLUENT INFLUENCE IN 2011 AND 2012

Golder Associates

PROJECT		PROJECT NO.	FILE No.	
DESIGN	GD	21 Nov. 2012	SCALE AS SHOWN	REV. 0
GIS	GI	01 Mar. 2013		
CHECK	LMB	19 Apr. 2013	FIGURE: 12.2-7	
REVIEW	PMC	18 Apr. 2013		

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Degree of Change between 2011 and 2012

Field conductivity measurements at the inlet and the outlet of each lake were higher in 2012 compared to 2011. The percent change ranged from 7% to 29% (Table 12.2-2).

Comparison to AEMP Benchmarks

Water quality data collected from the downstream lakes were compared to AEMP benchmarks, which refers to a list of generic water quality guidelines (e.g., CCME 1999 with updates; Health Canada 2012) and EAR benchmarks (De Beers 2002; Table 12.2-3). Most of the parameters measured at the downstream lakes in 2012 were below the protection of aquatic life and drinking water guidelines, with the exception of field pH and fluoride (Table 12.2-3). Field pH concentrations were below the lower range of the drinking water and aquatic life guidelines (pH of 6.5) at Inlet DSL1 and Inlet DSL2. Fluoride concentrations were above the aquatic life guidelines (0.12 milligrams per litre [mg/L]) at Inlet DSL1, DSL1-1, and DSL2-1. Concentrations of all measured parameters were below aquatic life and drinking water guidelines in Lac Capot Blanc in 2012.

12.2.4.3 Water Temperature Monitoring

Water temperature data collected during the open-water season (July to September 2012) from the temperature loggers set in DSL1, DSL2, and Lac Capot Blanc are presented in Figures 12.2-8 to 12.2-10.

The shallow site temperature logger in DSL1 was lost; Figure 12.2-8 presents the data for DSL2 and Lac Capot Blanc. The initial water temperature recorded in Lac Capot Blanc was warmer than DSL2 until mid-August where the temperatures in both lakes followed the same trend.

For the deep sample sites, the surface water temperatures showed a similar pattern for DSL1 and DSL2 from start to end with warmer temperatures in July and a general cooling off over the season (Figure 12.2-9), while Lac Capot Blanc water temperature was lower until early August where it warmed up and then followed the same cooling trend as DSL1 and DSL2.

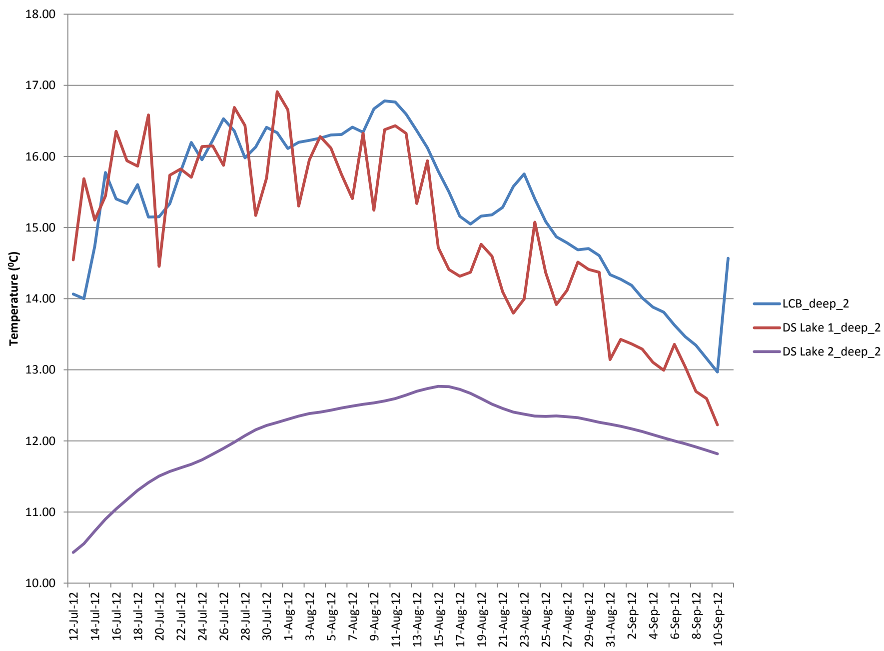
The deep temperature loggers for DSL1 and Lac Capot Blanc followed the same pattern and stable temperature range with a slight cooling off toward the end of the season while DSL2 was consistently colder (Figure 12.2-10). It can be assumed that the DSL2 temperature logger was sitting in the cooler substrate of the lake as the water temperature profile data from August 22, 2012 shows a stable temperature from surface to bottom of 15.2 degrees Celsius (°C) to 15.7ºC.

Figure 12.2-8 Water Temperature from Shallow Sample Sites (Total depth less than 1.0 m)

Note: Shallow thermograph in DSL1 was lost; therefore no temperature data are presented.

Figure 12.2-9 Water Temperature from Deep Sample Sites - Surface Measurements (Sample Depth 0.3 m)

Figure 12.2-10 Water Temperature from Deep Sample Sites - Deep Measurement (Sample Depth 1.0 m above Bottom Substrate)



Note: DSL2 data are very linear, assume that thermograph was in substrate and temperature data are not representative of actual water temperature.

12.2.4.4 Plankton Monitoring

Plankton and zooplankton samples collected during the 2012 open-water season from the three downstream lakes were archived for future analysis. Therefore, there are no results to report at this time for the downstream lakes.

Chlorophyll *a* concentrations in DSL1 (0.82), DSL2 (0.63), and Lac Capot Blanc (0.50) were lower than Snap Lake (1.02) but similar to Northeast Lake (0.74) during August 2012.

12.2.4.5 Sediment Quality

Results of the sediment chemistry analyses performed on the 2012 downstream lake sediment samples are reported in Table 12.2-4, along with information on comparisons to sediment quality guidelines (SQGs), field duplicate sample results, and comparisons to Snap Lake baseline normal ranges for each analyte. All results are presented on a dry weight basis.

A brief summary from the QA/QC review of these results is provided. Holding times were met for all analyses, except that total organic carbon (TOC) analyses were completed one or two days outside the 14-day recommended holding time. Target analytes were not detected in the method blanks, except that manganese was measured at the detection limit (DL; 1.0 milligrams per kilogram [mg/kg]) in one method blank. Laboratory duplicate analyses were not performed on downstream lake sediment sediments; instead they were performed on provisional reference Lake 13 sediment samples submitted to ALS in the same batch of samples (see Chapter 4 and Appendix 4A). A separate field duplicate sample was collected from the Lac Capot Blanc station (LCB-1); results for the original and field duplicate samples are provided in Table 12.2-4, along with the relative percent difference (RPD) for each analyte. The field duplicate RPDs ranged from less than 1% to 34%, and were less than 15% for most analytes. Results for laboratory reference materials were all within acceptable limits. No issues were identified with respect to sample analyses that would affect data quality.

Sediments from the three downstream lakes sampling stations consisted primarily of fine-grained material (silt and clay), 94% to 96% fines. The TOC concentrations ranged from 13.7% to 23.3%. Sediment particle size and TOC at these stations were similar to those measured at most stations in Snap Lake and Northeast Lake in previous AEMP programs. Concentrations of target analytes were either similar among the three stations or showed a net decrease with increasing distance downstream, except that concentrations of available phosphate, chromium, and uranium increased with distance downstream.

Table 12.2-3 Water Quality Results for Samples Collected from Downstream Lakes, 2012

Sample Point:	Units	AEMP Benchmark (Protection of Aquatic Life) [(a)]	Drinking Water[(a)]	Inlet DSL1	DSL1-1	Inlet DSL2	DSL2-1	Inlet 1 LCB	LCB-1	Outlet 1 LCB	Outlet 2 LCB
Sample Control Number:				2012-9006	2012-9011	2012-9007	2012-9012	2012-9008	2012-9013	2012-9009	2012-9010
Date Sampled (mm-dd- yyyy):				8/17/2012	8/22/2012	8/17/2012	8/22/2012	8/17/2012	8/23/2012	8/23/2012	8/26/2012
Field Parameters											
pH	-	6.5 to 9.0	6.5 to 8.5	**5.3**	6.7	**6.4**	7.1	6.6	7.2	7	7
Conductivity	µS/cm	-	-	377	276	263	244	240	35	36	26
Temperature	°C	-	-	13	15.2	13	15.3	12.1	15.4	16.2	14.9
Dissolved Oxygen	mg/L	-	-	9.9	10.8	10.4	10.8	10.3	10	10	10.3
Conventional Parameters											
Laboratory pH	-	-	-	7.4	7.4	7.4	7.4	7.3	7	7	6.9
Conductivity	µS/cm	-	-	389	285	276	253	254	34	35	25
Total Dissolved Solids	mg/L	-	-	286	191	202	164	189	12	22	18
Total Dissolved Solids, calculated (Lab)[(b)]	mg/L	-	-	187	139	131	123	120	17	18	16
Total Dissolved Solids, calculated (Standard Methods)[(b)]	mg/L	-	-	187	135	128	120	118	17	18	13
Total Suspended Solids	mg/L	-	-	<3	<3	<3	<3	<3	<3	<3	<3
Turbidity	NTU	-	-	0.33	0.36	0.3	0.31	0.24	0.48	0.61	0.44
Major Ions											
Bicarbonate, as HCO_3	mg/L	-	-	25	17	16	15	15	7	8	7
Calcium	mg/L	-	-	41	29	28	25	25	3	4	2
Carbonate, as CO_3	mg/L	-	-	<5	<5	<5	<5	<5	<5	<5	<5
Chloride	mg/L	120	<250	83	62	59	56	54	5	5	3
Fluoride	mg/L	0.12	1.5	**0.14**	**0.12**	0.11	**0.13**	0.12	0.07	0.07	0.09
Hardness, as $CaCO_3$	mg/L	-	-	125	88	83	77	77	12	13	9
Hydroxide, as OH	mg/L	-	-	<5	<5	<5	<5	<5	<5	<5	<5
Magnesium	mg/L	-	-	5.3	3.7	3.5	3.3	3.2	0.8	0.9	0.6
Potassium	mg/L	-	-	2	1.6	1.4	1.4	1.3	0.4	0.5	0.4
Reactive Silica, as SiO_2	µg/L	-	-	0.9	<0.5	0.7	0.5	0.6	<0.5	<0.5	<0.5
Sodium	mg/L	-	-	20	16	15	14	14	2	2	1
Sulphate	mg/L	-	-	16	11	11	10	10	2	2	1
Total Alkalinity, as $CaCO_3$	mg/L	-	-	20	14	13	13	12	6	7	6
Nutrients											
Dissolved Inorganic Phosphorus	mg-P/L	-	-	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
Dissolved Organic Phosphorus, calculated	mg-P/L	-	-	<0.0014	0.0021	<0.0014	0.0024	0.0017	0.002	0.0017	0.0015
Dissolved Phosphorus	mg-P/L	-	-	0.001	0.002	0.001	0.002	0.002	0.002	0.002	0.002
Nitrate, as N calculated	mg-N/L	2.93	10	1.46	0.65	0.55	0.43	0.4	<0.006	<0.006	<0.006

Table 12.2-3 Water Quality Results for Samples Collected from Downstream Lakes, 2012

Sample Point:	Units	AEMP Benchmark (Protection of Aquatic Life) [a]	Drinking Water[a]	Inlet DSL1	DSL1-1	Inlet DSL2	DSL2-1	Inlet 1 LCB	LCB-1	Outlet 1 LCB	Outlet 2 LCB
Sample Control Number:				**2012-9006**	**2012-9011**	**2012-9007**	**2012-9012**	**2012-9008**	**2012-9013**	**2012-9009**	**2012-9010**
Date Sampled (mm-dd- yyyy):				**8/17/2012**	**8/22/2012**	**8/17/2012**	**8/22/2012**	**8/17/2012**	**8/23/2012**	**8/23/2012**	**8/26/2012**
Nitrite, as N	mg-N/L	0.06	1	0.008	0.006	0.003	0.005	<0.002	<0.002	<0.002	<0.002
Nitrate/Nitrite, as N	mg-N/L	-	-	1.47	0.66	0.56	0.43	0.4	<0.006	<0.006	<0.006
Ortho-Phosphate, as P	mg-P/L	-	-	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
Total Ammonia, as N	mg-N/L	2.1 to 9 [c]	-	0.019	<0.005	0.006	<0.005	<0.005	<0.005	<0.005	<0.005
Total Inorganic Phosphorus	mg-P/L	-	-	0.002	0.002	0.001	0.002	0.002	0.005	0.002	0.002
Total Kjeldahl Nitrogen	mg/L	-	-	0.24	0.31	0.15	0.26	0.11	0.14	0.18	0.2
Total Organic Carbon	mg/L	-	-	3	3.6	3	3.4	2.8	3.7	4.2	2.9
Total Organic Phosphorus, calculated	mg-P/L	-	-	0.002	0.002	0.002	0.002	<0.0014	<0.0014	0.002	0.002
Total Phosphorus	mg-P/L	-	-	0.003	0.004	0.004	0.004	0.003	0.003	0.004	0.004
Total Metals and Metalloids											
Aluminum	µg/L	100 [d]	-	4	6.6	5.3	8.1	3.5	2.8	3.8	5.3
Antimony	µg/L	-	-	0.04	0.03	<0.02	0.02	<0.02	<0.02	<0.02	<0.02
Arsenic	µg/L	5	25	0.08	0.08	0.08	0.08	0.07	0.07	0.07	0.07
Barium	µg/L	-	1,000	15	12	12	11	10	2.9	3	2.9
Beryllium	µg/L	-	-	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
Boron	µg/L	1,500	5,000	31	21	20	18	17	3	3	4
Cadmium	µg/L	0.36	5	0.005	0.002	0.003	<0.002	0.002	<0.002	<0.002	<0.002
Chromium	µg/L	1	50	<0.06	<0.06	<0.06	<0.06	<0.06	<0.06	<0.06	<0.06
Hexavalent Chromium	µg/L	2.1	-	<1	<1	<1	<1	<1	<1	<1	<1
Cobalt	µg/L	-	-	0.01	0.02	0.01	0.01	<0.01	<0.01	<0.01	0.01
Copper	µg/L	7.9	1,000	0.4	0.4	0.4	0.4	0.4	0.3	0.2	0.3
Iron	µg/L	300	-	12	17	21	23	17	8	14	13
Lead	µg/L	1.0 to 4.2 [e]	10	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
Lithium	µg/L	-	-	6.6	5	4.4	4.2	3.7	1.2	1.2	0.7
Manganese	µg/L	-	-	2.8	5.3	3.3	3.3	2.6	5.7	5.3	5.1
Mercury	µg/L	0.026	1	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005	0.0016	<0.0005	<0.0005
Molybdenum	µg/L	73	-	0.79	0.35	0.3	0.26	0.23	<0.05	<0.05	0.12
Nickel	µg/L	25 to 113 [e]	-	0.44	0.22	0.21	0.15	0.14	0.07	0.08	0.07
Selenium	µg/L	1	10	<0.04	<0.04	<0.04	<0.04	<0.04	<0.04	<0.04	<0.04
Silver	µg/L	0.1	-	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005
Strontium	µg/L	-	-	500	365	340	312	299	30.1	31.7	17.8
Thallium	µg/L	0.8	-	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
Titanium	µg/L	-	-	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1

Table 12.2-3 Water Quality Results for Samples Collected from Downstream Lakes, 2012

Sample Point:	Units	AEMP Benchmark (Protection of Aquatic Life) (a)	Drinking Water(a)	Inlet DSL1	DSL1-1	Inlet DSL2	DSL2-1	Inlet 1 LCB	LCB-1	Outlet 1 LCB	Outlet 2 LCB
Sample Control Number:				2012-9006	2012-9011	2012-9007	2012-9012	2012-9008	2012-9013	2012-9009	2012-9010
Date Sampled (mm-dd- yyyy):				8/17/2012	8/22/2012	8/17/2012	8/22/2012	8/17/2012	8/23/2012	8/23/2012	8/26/2012
Uranium	µg/L	15	20	0.12	0.18	0.17	0.31	0.15	0.05	0.05	0.05
Vanadium	µg/L	-	-	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05
Zinc	µg/L	30	-	<0.8	<0.8	1	<0.8	1.6	<0.8	0.8	1.4

Note: **Bold** values are above the relevant aquatic life guidelines. Underlined values are above the relevant drinking water guidelines.

(a) AEMP benchmarks include: aquatic life guidelines from the Canadian Council of Ministers of the Environment (CCME) (1999 with updates to 2012) 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). Drinking water guidelines are from Health Canada (2012).

(b) "Total dissolved solids, calculated (Lab)" refers to laboratory-calculated total dissolved solids concentrations using a formula inconsistent with Method 1030 E in the Standard Methods for the Examination of Water and Wastewater, 21st Edition (APHA 2005). Total dissolved solids, calculated (Standard Methods) concentrations were calculated using formula outlined in APHA (2005). Refer to Appendix 3-A, Section 3A.4.2.2.3 for further details.

(c) The ammonia guideline is pH and water temperature dependent. The guideline shown is based on a range of laboratory pH from 6.9 to 7.4 and a range of water temperature from 12.1 to 16.2°C. The guideline was calculated based on an individual pH and water temperature for each sample with the final value expressed as total ammonia as nitrogen.

(d) Aluminum guideline is pH dependent. The guideline shown here is based on a range of pH from 6.9 to 7.4. The guideline was calculated based on the individual pH for each sample.

(e) Lead and nickel guidelines are hardness dependent and were based on a range of hardness (as calcium carbonate) of 8.5 to 125 mg/L. The guideline was calculated based on the individual hardness for each sample.

HCO_3 = bicarbonate; $CaCO_3$ = calcium carbonate; CO_3 = carbonate; OH = hydroxide; SiO_2 = silicate; N = nitrogen; P = phosphorus; °C = degrees Celsius; µS/cm = microSiemens per centimetre; mg/L= milligrams per litre; µg/L= micrograms per litre; mg-N/L = milligrams as nitrogen per litre; mg-P/L = milligrams as phosphorus per litre; NTU = nephelometric turbidity units; - = not applicable; <= less than the detection limit; mm-dd- yyyy = month-day- year; DSL1 = first lake downstream of Snap Lake; DSL2= second lake downstream of Snap Lake; LCB = Lac Capot Blanc.

Table 12.2-4 Sediment Quality Results for Samples Collected from Downstream Lakes, 2012

Station Name	Units (dry wt)	Detection Limits (DLs)	CCME SQGs[a]		DSL1-1	DSL2-1	LCB-1	LCB-1 (field dup)	LCB-1 RPD (%)	Snap Lake Normal Range (Mean ± 2SD)	Comparison to Snap Lake Normal Ranges
Sample ID (Golder SCN)			ISQG	PEL	2012-9070	2012-9071	2012-9072	2012-9073			
Physical											
Fines (Silt + Clay)	%	0.1	-	-	96.5	95.9	94.3	94.5	0.2	87.3 - 100.3	Within
Total Organic Carbon	%	0.1	-	-	17	23.3	13.7	14	2.2	9.9 - 29.1	Within
Nutrients											
Available Ammonium-N	mg/kg	2.4	-	-	44.1	28.1	30	28	6.9	13.9 - 87.3	Within
Available Nitrate-N	mg/kg	4	-	-	<8.0	<8.0	<8.0	<8.0	n/a	0 - 68.8	Within
Total Kjeldahl Nitrogen (TKN)	%	0.02	-	-	1.42	1.8	1.11	1.1	0.9	0.70 - 2.17	Within
Total Nitrogen	%	0.02	-	-	1.43	1.83	1.06	1.09	2.8	0.85 - 2.21	Within
Available Phosphate-P	mg/kg	4	-	-	7	<4.0	30.5	21.6	34.2	0 - 38.8	Within
Available Potassium	mg/kg	20	-	-	200	96	100	101	1	27.7 - 156	Above (DSL1-1)
Available Sulfate-S	mg/kg	3	-	-	154	47.9	11.4	11.3	0.9	0 - 233	Within
Metals											
Aluminum (Al)	mg/kg	50	-	-	11,700	10,100	9,840	10,100	2.6	8,539 - 21,326	Within
Antimony (Sb)	mg/kg	0.1	-	-	0.42	0.18	0.16	0.13	20.7	0.10 - 0.10	Above (all stations)
Arsenic (As)	mg/kg	0.1	5.9	17	3.44	1.51	1.14	1.22	6.8	1.24 - 4.41	Below (LCB-1)
Barium (Ba)	mg/kg	0.5	-	-	74	55.3	78.4	80.8	3	0 - 834	Within
Beryllium (Be)	mg/kg	0.2	-	-	1.23	1.55	0.77	0.88	13.3	0.51 - 1.44	Above (DSL2-1)
Bismuth (Bi)	mg/kg	0.2	-	-	1.36	0.64	0.32	0.31	3.2	0.40 - 0.65	Above (DSL1-1)
Boron (B)	mg/kg	2	-	-	17	8.6	12.2	12.3	0.8	2.8 - 23.4	Within
Cadmium (Cd)	mg/kg	0.1	0.6	3.5	0.4	0.58	0.46	0.48	4.3	0.34 - 1.05	Within
Calcium (Ca)	mg/kg	100	-	-	5,440	5,920	2,950	2,990	1.3	2,924 - 5,510	Above (DSL2-1)
Cesium (Cs)	mg/kg	0.1	-	-	2.1	1.59	1.96	2.01	2.5	0.48 - 3.29	Within
Chromium (Cr)	mg/kg	0.5	37.3	90	30.3	**46.3**	**53.2**	**60.5**	12.8	17.6 - 55.0	Above (LCB-1 dup)
Cobalt (Co)	mg/kg	0.1	-	-	17.5	26.8	6.29	6.54	3.9	6.6 - 16.6	Below (LCB-1)
Copper (Cu)	mg/kg	0.5	35.7	197	**83.4**	**103**	**60.7**	**62.6**	3.1	75 - 124	Below (LCB-1)
Iron (Fe)	mg/kg	200	-	-	61,700	50,200	16,300	16,500	1.2	4,874 - 44,426	Above (DSL1-1, DSL2-1)
Lead (Pb)	mg/kg	0.5	35	91.3	8.75	7.93	6.77	6.99	3.2	2.4 - 8.6	Above (DSL1-1)
Lithium (Li)	mg/kg	0.5	-	-	23.5	17.7	22.2	22.9	3.1	3.3 - 38.7	Within
Magnesium (Mg)	mg/kg	20	-	-	3,340	2,260	3,370	3,460	2.6	591 - 6,854	Within
Manganese (Mn)	mg/kg	1	-	-	274	173	161	160	0.6	96 - 478	Within
Mercury (Hg)	mg/kg	0.05	0.17	0.49	0.08	<0.050	<0.050	<0.050	NA	0.05 - 0.06	Above (DSL1-1)
Molybdenum (Mo)	mg/kg	0.1	-	-	11.4	13.6	4.89	4.97	1.6	1.9 - 17.3	Within
Nickel (Ni)	mg/kg	0.5	-	-	30	52.7	33	37	11.4	26.6 - 56.6	Within
Phosphorus (P)	mg/kg	100	-	-	1,210	840	700	710	1.4	594 - 2,994	Within
Potassium (K)	mg/kg	100	-	-	1,700	990	1,760	1,780	1.1	0 - 3,650	Within
Rubidium (Rb)	mg/kg	1	-	-	14.4	10.7	15.3	15.6	1.9	0.6 - 26.7	Within

Table 12.2-4 Sediment Quality Results for Samples Collected from Downstream Lakes, 2012

Station Name	Units (dry wt)	Detection Limits (DLs)	CCME SQGs[(a)]		DSL1-1	DSL2-1	LCB-1	LCB-1 (field dup)	LCB-1 RPD (%)	Snap Lake Normal Range (Mean ± 2SD)	Comparison to Snap Lake Normal Ranges
Sample ID (Golder SCN)			ISQG	PEL	2012-9070	2012-9071	2012-9072	2012-9073			
Selenium (Se)	mg/kg	0.1	-	-	1.54	1.91	1.06	1.13	6.4	0.10 - 0.10	Above (all stations)
Silver (Ag)	mg/kg	0.2	-	-	0.21	<0.20	<0.20	<0.20	NA	0.20 - 0.20	Above (DSL1-1)
Sodium (Na)	mg/kg	100	-	-	410	410	150	160	6.5	139 - 345	Above (DSL1-1, DSL2-1)
Strontium (Sr)	mg/kg	1	-	-	69.7	83.3	28.3	29	2.4	15.7 - 39.1	Above (DSL1-1,DSL2-1)
Thallium (Tl)	mg/kg	0.05	-	-	0.106	0.09	0.086	0.091	5.6	0 - 0.41	Within
Tin (Sn)	mg/kg	2	-	-	<2.0	<2.0	<2.0	<2.0	NA	2.00 - 2.00	Within
Titanium (Ti)	mg/kg	1	-	-	291	252	268	271	1.1	98 - 822	Within
Uranium (U)	mg/kg	0.05	-	-	20.5	39.4	28.3	28.9	2.1	3.5 - 14.6	Above (all stations)
Vanadium (V)	mg/kg	0.2	-	-	29.6	23.6	27.3	28	2.5	16.6 - 46.4	Within
Zinc (Zn)	mg/kg	5	123	315	**129**	**141**	96	101	5.1	72 - 298	Within

Note: **Bold** values are above the CCME ISQG.

% = percent; mg/kg = milligrams per kilogram; SD = standard deviation; <= less than the detection limit; dup = duplicate; wt = weight; DL = detection limit; RPD = relative percent difference; Golder = Golder Associates Ltd.; SCN = sample control number; ID = identifier; CCME = Canadian Council of Ministers of the Environment; SQG = sediment quality guideline; ISQG = interim sediment quality guideline; PEL = probable effect level; DSL1 = first lake downstream of Snap Lake; DSL2 = second lake downstream of Snap Lake; LCB = Lac Capot Blanc; - = not applicable; n/a = not available.

Sediment quality data were compared to the Interim Sediment Quality Guidelines (ISQGs) and Probable Effect Levels (PEL) (CCME 1999 with updates), which were available for seven metals analyzed in Snap Lake and Northeast Lake sediments (Table 12.2-4). The ISQG is the concentration of a substance below which an adverse effect on aquatic life is unlikely, and 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. None of the metals concentrations were above PELs but concentrations of three metals were above their respective ISQGs: chromium (Stations DSL2-1 and LCB-1); copper (all three stations); and, zinc (Stations DSL1-1 and DSL2-1). Concentrations of these metals have also been above their respective ISQGs in sediments from Snap Lake and Northeast Lake in AEMP monitoring, reflecting the natural enrichment of the region.

Sediment quality data for the downstream lakes stations were compared to Snap Lake normal ranges (Table 12.2-4), which were derived from 2004 baseline monitoring data (or from the first year that the analyte was monitored if bottom conductivity data showed that stations had not yet been exposed to treated effluent discharge). Concentrations of available potassium, antimony, beryllium, bismuth, calcium, chromium, iron, lead, mercury, selenium, silver, sodium, strontium, and uranium were above Snap Lake normal ranges at one or more downstream lakes stations. Concentrations of arsenic, cobalt, and copper were below Snap Lake normal ranges at Station LCB-1 in Lac Capot Blanc.

12.2.4.6 Benthic Invertebrates

Supporting Environmental Variables

During benthic invertebrate sampling, conductivity levels were higher at DSL1-1 (277 µS/cm), and DSL2-1 (243 µS/cm), compared to LCB-1 (35 µS/cm) (Table 12.2-5), reflecting the presence of treated effluent at DSL1-1 and DSL2-1. Conductivity was only slightly above background at LCB-1. Other field measured water quality parameters, including DO, water temperature, and pH, were similar among the three downstream lakes (Table 12.2-5).

Table 12.2-5 Station Locations and Field Water Quality Parameters Measured in DSL1, DSL2, and Lac Capot Blanc, August 2012

Station	Date	UTM Coordinates		Sample Depth	Water Temperature	Dissolved Oxygen	Dissolved Oxygen	Conductivity	pH
		Easting	Northing	(m)	(°C)	(mg/L)	(%)	(µS/cm)	
DSL1-1	22-Aug-12	513403	7054940	14	15	10.1	100	277	6.9
DSL2-1	22-Aug-12	515197	7053418	6	15.2	10.9	108	243	7.1
LCB-1	23-Aug-12	518411	7053352	13	15.3	10	100	35	7.2

DSL1 = first lake downstream of Snap Lake; DSL2= second lake downstream of Snap Lake, LCB = Lac Capot Blanc.
Sample depth indicates depth that water quality readings were collected and are not the maximum depth at the station.
UTM = Universal Transverse Mercator; North American Datum (NAD) 83, Zone 12V.
m = metre; °C = degrees Celsius; mg/L = milligrams per litre; % = percent; µS/cm = microSiemens per centimetre.

Water depth at the benthic invertebrate sampling locations was similar at DSL1-1 and LCB-1 (14.5 and 14.8 m) but was shallower at DSL2-1 (6.9 m; Table 12.2-6). The inorganic content of bottom sediments was similar among the three downstream lakes, and consisted primarily of silt with smaller amounts of sand and clay. Bottom sediments were composed of between 94% and 97% fines (silt + clay), indicating a low range of variation in sediment particle size among the sample locations. The TOC content of sediments was generally high for oligotrophic lakes, and varied between 14% (LCB-1) and 23% (DSL2-1).

Table 12.2-6 Water Depth, Sediment Organic Carbon and Inorganic Particle Size DSL1, DSL2, and Lac Capot Blanc, August 2012

Lake	Water Depth (m)	Total Organic Carbon (%)	Sediment Particle Size				
			Gravel (%)	Sand (%)	Silt (%)	Clay (%)	Fines (Silt + Clay) (%)
DSL1-1	14.5	17	<0.1	3	88	9	97
DSL2-1	6.9	23	<0.1	4	88	8	96
LCB-1	14.8	14	<0.1	6	79	16	94

m = metre; % = percent; DSL1 = first lake downstream of Snap Lake; DSL2= second lake downstream of Snap Lake; LCB = Lac Capot Blanc.

Benthic Invertebrate Community Summary Variables

Benthic invertebrate density was variable among the three downstream lakes, with mean values of 6,444 org/m² at DSL1-1, 359 org/m² at DSL2-1, and 144 org/m² at LCB-1 (Appendix 12B.6, Table 12B.6-1). The dominant benthic taxa at DSL1-1 and DSL2-1 were the Chironomidae (midges), which accounted for 63% and 54% (respectively) of the total density (Table 12.2-8). Pisidiidae (fingernail clams) were the second most abundant taxa at DSL1-1 and DSL2-1, contributing 31% and 26% (respectively) of the total density. Oligochaeta (aquatic worms) represented 16% of the total abundance at DSL2-1 but were less common at DSL1-1 (2%). The majority of the Chironomidae density at DSL1-1 and DSL2-1 consisted of the Chironomini and Tanytarsini tribes.

Table 12.2-7 Benthic Invertebrate Summary Variables in DSL1, DSL2, and Lac Capot Blanc, August 2012

Station	Total Density (no./m²)		Total Richness (taxa/station)	Simpson's Diversity Index	Evenness
	Mean	Standard Error			
DSL1-1	6,444	1,387	22	0.81	0.24
DSL2-1	359	169	10	0.81	0.53
LCB-1	144	78	9	0.86	0.77

no = number, m^2 = square metre. DSL1 = first lake downstream of Snap Lake; DSL2= second lake downstream of Snap Lake; LCB = Lac Capot Blanc.

Table 12.2-8 Relative Densities of Dominant Taxa in Lake DSL1, Lake DSL2, and Lac Capot Blanc, August 2012

Taxon	DSL1 (%)	DSL2 (%)	Lac Capot Blanc (%)
Pisidiidae	30.8	26	45
Tanypodinae	4.6	6	0
Chironomini	30.7	12	20
Tanytarsini	27.2	36	5
Orthocladiinae	0.1	0	0
Other Chironomidae	0.2	0	0
Other	6.5	20	30
Total	**100**	**100**	**100**
Total Chironomidae	62.8	54	25

% = percent. DSL1 = first lake downstream of Snap Lake; DSL2= second lake downstream of Snap Lake.

The dominant taxa at LCB-1 were the Pisidiidae (45%; Table 12.2-8). The Chironomidae were the second most common taxa in the lake, accounting for 25% of the total abundance. The majority of the Chironomidae density at LBC-1 consisted of the Chironomini tribe. Oligochaeta and Gastropoda (snails) were also abundant at LCB-1, each contributing 15% of the total density.

Richness values in the downstream lakes were within the expected range for lake habitats in the sub-Arctic region (Table 12.2-7). Total richness was about two times higher at DSL1-1 (22 taxa/station), compared to DSL2-1 (10 taxa/station) and LCB-1 (9 taxa/station). Amphipoda (amphipods), Nematoda (roundworms), Hydracarina (water mites), and Ostracoda (ostracods) were only present at DSL1-1 (Table 12.2-9). All four major midge groups were represented at DSL1-1, compared to 3 groups at DSL2-1 and 2 groups at LCB-1.

Simpson's diversity values in the downstream lakes varied between 0.81 and 0.86, indicating a high level of benthic invertebrate diversity at the stations sampled. Evenness was variable among the three waterbodies, with low to moderate values observed at DSL1-1 (0.24) and DSL2-1 (0.53), and high evenness observed at LCB-1 (0.77; Table 12.2-7). The low evenness value for DSL1-1 indicated that a few taxa account for the majority of the total density at this station. The moderate and high evenness values likely result from the low densities at these stations and the fact that the few taxa present are represented by only a few individuals, which leads to higher evenness values.

Table 12.2-9 Presence/Absence of Benthic Invertebrate Taxa in DSL1, DSL2, and Lac Capot Blanc, August 2012

Major Taxon	Family	Subfamily/ Tribe	Genus/Species	DSL1-1	DSL2-1	LCB-1	Area Total
Nematoda	-	-	-	X			X
Oligochaeta	Lumbriculidae	-	*Lumbriculus*			X	X
	Naididae	Naidinae	-	X			X
		Tubificinae	-	X	X	X	X
Hydracarina	-	-	-	X			X
Ostracoda	-	-	-	X			X
Amphipoda	Talitridae	-	*Hyalella azteca*	X			X
Gastropoda	Valvatidae	-	*Valvata sincera*	X	X	X	X
Pelecypoda	Pisidiidae	-	*Pisidium*	X	X	X	X
		-	*Sphaerium*	X	X	X	X
		-	*Pisidium / Sphaerium*	X	X	X	X
Diptera	Chironomidae	Tanypodinae	*Procladius*	X	X		X
		Chironomini	*Cladopelma*	X	X		X
			Cryptochironomus	X	X		X
			Microtendipes	X		X	X
			Pagastiella	X	X		X
			Sergentia	X			X
			Stictochironomus	X		X	X
		Tanytarsini	*Cladotanytarsus*	X			X
			Corynocera	X	X	X	X
			Tanytarsus	X			X
		Orthocladiinae	*Psectrocladius*	X			X
		Diamesinae	*Protanypus*	X			X

- = not identified to this taxonomic level. X = present; blank cell = absent. DSL1 = first lake downstream of Snap Lake; DSL2 = second lake downstream of Snap Lake; LCB = Lac Capot Blanc.

12.2.5 Summary and Conclusions

Key Question 1: What is the spatial extent of the treated effluent plume downstream of Snap Lake?

Evidence of the treated effluent was detected throughout DSL1 and DSL2, and near the inlet of Lac Capot Blanc in 2012. Treated effluent extended approximately 650 m from the inlet of Lac Capot Blanc and approximately 5.8 km downstream from Snap Lake's outlet in 2012 (Figure 12.2-7). Based on the 2012 conductivity values, the area influenced by treated effluent has increased since 2011, when it decreased to background levels within 50 m of the inlet of Lac Capot Blanc (De Beers 2012a). The treated effluent discharged from the Snap Lake Mine tends to sink to the bottom as it enters Lac Capot Blanc.

Key Question 2: What are the current water and sediment quality characteristics in the three downstream lakes?

Based on the field measurements collected in 2012 (Table 12.2-2), the water in downstream lakes DSL1 and DSL2, and Lac Capot Blanc was well oxygenated, and varied from slightly acidic to slightly alkaline. Field pH measurements at Inlet DSL1 and Inlet DSL2 were outside the lower range of the drinking water and protection of aquatic life guidelines (pH of 6.5).

Concentrations of TDS, nitrate, and major ions were elevated in DSL1 and DSL2 (Table 12.2-2) and decreased at LCB-1 in Lac Capot Blanc in 2012 (Table 12.2-2), indicating that the influence of the treated effluent extends beyond the inlet of Lac Capot Blanc. Most parameters in downstream lakes in 2012 were below guidelines for the protection of aquatic life and drinking water supply, with the exception of fluoride (Table 12.2-2). Fluoride concentrations were above the aquatic life guidelines (0.12 mg/L) at three stations (Inlet DSL1, DSL1-1, and DSL2-1). The same decreasing trend was also observed in barium, boron, molybdenum, rubidium, and strontium, which are also characteristic of the treated effluent (Table 12.2-2). Total metal concentrations were below aquatic life and drinking water guidelines in DSL1, DSL2, and Lac Capot Blanc in 2012. The results indicate that the influence from the Mine was reduced as total watershed areas and inflows to the lakes increased.

Based on the field data collected in 2012 (Table 12.2-4), bottom sediments at stations sampled in the three downstream lakes consisted primarily of fine-grained material (silt and clay), 94% to 96% fines. The TOC concentrations ranged from 13.7% to 23.3%. Sediment particle size and TOC at these stations were similar to those measured at most stations in Snap Lake and Northeast Lake in previous AEMP programs. Concentrations of target analytes were either similar among the three stations or showed a net decrease with increasing distance downstream, except that concentrations of available phosphate, chromium, and uranium increased with distance downstream.

None of the metals concentrations were above PELs but concentrations of three metals were above their respective ISQGs: chromium (Stations DSL2-1 and LCB-1); copper (all three stations); and, zinc (Stations DSL1-1 and DSL2-1). Concentrations of these metals have also been above their respective ISQGs in sediments from Snap Lake and Northeast Lake in AEMP monitoring, reflecting the natural enrichment of the region.

Concentrations of available potassium, antimony, beryllium, bismuth, calcium, chromium, iron, lead, mercury, selenium, silver, sodium, strontium, and uranium were above Snap Lake normal ranges at one or more downstream lakes stations. Concentrations of arsenic, cobalt, and copper were below Snap Lake normal ranges at Station LCB-1 in Lac Capot Blanc.

Key Question 3: What is the current benthic invertebrate community composition in the three downstream lakes?

The benthic invertebrate community was characterized by low (DSL2-1 and LCB-1) to moderate (DSL1-1) density in the downstream lakes. The benthic invertebrate community in the downstream lakes was dominated by midges (DSL1-1 and DSL2-1) and fingernail clams (LCB-1). Total richness was variable, ranging from low (DSL2-1 and LCB-1) to moderate (DSL1-1). The Simpson's diversity index for the benthic invertebrate community was high and evenness values were low indicating a diverse benthic invertebrate community with a few taxa accounting for the majority of the total density in each lake. The benthic invertebrate community summary variables in the downstream lakes were within the normal range for Northeast Lake and were similar to ranges observed in Snap Lake in 2011. Overall, the benthic invertebrate community is as expected for lakes in the sub-Arctic region.

12.2.6 Recommendations

Treated effluent is becoming evenly mixed throughout the main body of Snap Lake and, as predicted, is now present in lakes immediately downstream of Snap Lake. Based on the 2011 reconnaissance survey and 2012 sampling program, concentrations of Mine-related constituents reached background within approximately 6 km downstream of Snap Lake. In the EAR, concentrations were predicted to reach near background concentrations approximately 44 km downstream of Snap Lake at the end of operations.

It is recommended that monitoring in the downstream lakes (DSL1 and DSL2) and Lac Capot Blanc be continued to evaluate the dispersion of treated effluent associated with Snap Lake Mine discharge. The 2013 study design, as outlined in the 2013 AEMP Design Plan (De Beers 2012b) includes sampling of downstream lakes. The 2013 downstream sampling program will gather information on the downstream spatial extent of the treated effluent plume and on water and sediment quality. Examination of the 2013 water and sediment data will be used to determine future monitoring programs that may be affected by the treated effluent including (fish, plankton and benthic invertebrates).

It is also recommended that the downstream water quality predictions be revisited, so that mixing and other processes can be considered. For the EAR and Water Licence renewal, an Excel-based mixing model was used to calculate TDS concentrations in lakes downstream of Snap Lake. The model was steady-state, so it conservatively represented a snapshot in time assuming that peak TDS concentrations remained in Snap Lake indefinitely. The model did not compute mixing patterns within each of the lakes, or provide time-varying estimates of concentrations at particular nodes, nor did it account for time of travel through the Lockhart River system.

More rigorous predictions (including timing and movement of the treated effluent plume) would support the downstream lakes monitoring program development. Modelling results could then be considered during the individual selection of new monitoring stations. Therefore, it is suggested that a hydrodynamic downstream model be developed using the three dimensional (3-D) hydrodynamic and water quality model that was used to model water quality in Snap Lake (Generalized Environmental Modelling System for Surfacewaters; GEMSS®). The model would be used to predict TDS concentrations at various points in the downstream lakes including near the inlet and outlet, in deep pockets, and as whole-lake averages. The initial focus of the model would be the first three downstream lakes (i.e., DSL1, DSL2, and Lac Capot Blanc), but could be expanded in future as necessary and appropriate.

The temperature logger program should install temperature loggers earlier in the year to capture spring temperature variations. Redundancy should be built into the temperature logger program to verify data and reduce the potential loss of data from field error and equipment failure. It is recommended that one of the shallow site temperature loggers be installed as close to the inlet stream as possible for modelling.

Additional bathymetric surveys of the east basin and southern portions of Lac Capot Blanc to fill in data gaps are also recommended to provide more detailed bathymetry maps of the lake.

12.2.7 References

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12.3 SUITABILITY OF LAKE 13 AS A REFERENCE LAKE SPECIAL STUDY

12.3.1 Introduction

In 2006, Northeast Lake was approved by the Mackenzie Valley Land and Water Board (MVLWB) as the reference lake for the Snap Lake Aquatics Effects Monitoring Program (AEMP) following a survey of potential reference lakes (Golder 2005). Thirty-three lakes were reviewed on the basis of size, shape, distance from the Mine, and physical characteristics. From this review, five of the lakes thought to be most similar to Snap Lake were selected for further evaluation as possible reference lakes. Site-specific information was collected for these five lakes. The lake identified as most similar to Snap Lake was Northeast Lake. Northeast Lake was approved by the MVLWB as the reference lake for the Snap Lake AEMP in 2006.

In 2012, the Aquatic Effects Re-evaluation Report recommended the addition of a second reference lake for the 2013 AEMP (De Beers 2012a). Due to the inherent natural differences in lakes within the region, particularly in nutrient concentrations, a multiple reference lake design was recommended to assist in establishing a more regional context for Snap Lake. The intent of a second reference lake is to further understand and bound natural variability, and to assist in determining whether changes in Snap Lake are natural or Mine-related.

In the 2013 AEMP Design Plan (De Beers 2012b) it was proposed that Lake 13 be added to the AEMP as a second reference lake. This lake was identified as the ‘second most similar’ lake to Snap Lake in the original survey of potential reference lakes, despite the fact that the winter road to the Mine crosses part of the lake (Golder 2005). Based on the 2005 reference lake survey, it is unlikely that a more similar lake (e.g., similar habitat and bathymetry to Snap Lake) can be found (Golder 2005). Thus, Lake 13 was selected provisionally as the second reference lake, and was included in the 2012 AEMP. Lake 13 was subsequently approved as the second reference lake by the Board in March 2013, with the condition that winter road monitoring be included in the final 2013 AEMP Design Plan (MVLWB 2013).

12.3.1.1 Objective

In 2012, as part of a Special Study, De Beers collected water, sediment, plankton, benthic invertebrate community, fish health, and fish tissue data from Lake 13. The objective of this Special Study was to update the 2005 information on Lake 13 to further assess the comparability of Lake 13 to Snap Lake and Northeast Lake.

12.3.2 Methods

Detailed field methods are presented in each section of this 2012 AEMP Annual Report (as summarized in Table 12.3-1). Results were compared to the results from the 2005 reference lake survey (Golder 2005), as well as to the 2012 data collection on Northeast Lake and Snap Lake. The 2012 field data were then reviewed against the 2005 selection criteria for reference lakes and any discrepancies were noted.

Table 12.3-1 Summary of 2012 Data Collected From Lake 13

Component	Stations	Frequency and Timing	Data Type	Location in Report
Physical	Whole lake	Bathymetry conducted once in summer, water temperature recorded from July to September	Bathymetry and water temperature	Section 3 (Figure 3.3); Section 2
Water	5 stations	Sampled monthly during the open-water season	Metals, nutrients, and major ions	Section 3.2
Sediment	5 stations	Sampled once in fall	Chemistry including metals, TOC, and particle size	Section 4.2
Plankton	1 station	Sampled once in August	Phytoplankton community composition, zooplankton biomass and abundance, and zooplankton community composition	Section 5.2
Benthic Invertebrates	5 stations	Sampled once in fall	Benthic invertebrate biomass and abundance, and community composition	Section 6.2
Fish health	Whole lake, main basin	Sampled once in July	Age, length, weight, gonad and liver size of Lake Chub	Section 7.2
Fish Tissue Chemistry	Whole lake, main basin	Sampled once in July	Metals, major ions, lipid concentration in Lake Chub carcass (flesh and bones, but not viscera, liver or stomach)	Section 9.2

TOC = total organic carbon

12.3.3 Results

The physical characteristics of Lake 13 remain similar to Snap Lake in that Lake 13 is a multi-basin headwater lake; however, the amount of shoreline habitat in Snap Lake is greater than in Lake 13, largely due to the narrow shape of the Northwest Arm (Golder 2005). The updated bathymetry of Lake 13 (see Figure 3.3), completed in 2012, confirms the coarse-level bathymetry from 2005.

Water temperature was measured during the open-water season for the first time in 2012. Water temperature data collected from the temperature loggers for Snap Lake, Northeast Lake, and Lake 13 are presented in Figures 2-7 to 2-9. The shallow temperature loggers (Figure 2-7) followed a similar trend in Snap Lake, Northeast Lake, and Lake 13 through the late spring and summer. The water temperature in Northeast Lake was lower than the other lakes during the first 10 days of measurement (between July 10 and 20, 2012). Lake 13 temperature measurements were consistently low (i.e., around 11 degrees Celsius [°C]) early in the season and then sharply increased in mid-August to 14.5°C (Figure 2-9). In general, Snap Lake and the reference lakes were similar; however, Lake 13 took longer to warm up in summer.

12.3.3.1 Water Quality

Water quality data collected from Lake 13 in 2005 and 2012 were similar (Appendix 3C; Golder 2005). Maximum concentrations for most parameters in 2012 were within the range measured in 2005, with the exception of total phosphorus, boron, manganese, molybdenum, strontium, and zinc, which are discussed in more detail below. Concentrations of total dissolved solids and its component ions, were low in Lake 13 in both years, and were similar to concentrations in Snap Lake prior to treated effluent discharge. In 2012, Lake 13 was well oxygenated with low turbidity and had a mildly acidic pH, similar to 2005. Concentrations of nitrate, nitrite, and ammonia were below detection limits (DLs) in 2012. Concentrations of total Kjeldahl nitrogen were above the DL, but within the 2005 range. For phosphorus nutrients, concentrations of dissolved and inorganic phosphorus were below DLs in 2012. The maximum total phosphorus concentration in 2012 was 0.005 milligram per litre (mg/L), compared to a maximum of 0.003 mg/L in 2005.

Concentrations of most total metals were consistent between 2005 and 2012 in Lake 13. For eight metals (i.e., silver, uranium, beryllium, bismuth, cobalt, lead, selenium and thallium), detection limits improved, so concentrations were recorded as lower in 2012. Total strontium and zinc concentrations were slightly higher in 2012 compared to 2005 concentrations. Maximum total strontium and zinc concentrations were 10 microgram per litre (µg/L) and 1.9 µg/L, in 2012, respectively, compared to 9 µg/L and 1.0 µg/L in 2005, respectively. Concentrations of total molybdenum, boron, and manganese in Lake 13, were notably different in 2012 compared to 2005. For molybdenum, all samples were below the DL (i.e., 0.05 µg/L), with the exception of one sample collected at LK13-01 with a concentration of 0.14 µg/L. Follow-up sampling will determine whether molybdenum concentrations continue to be elevated at this location or whether this single result was a potential isolated error during the August sampling event. For comparison, maximum molybdenum concentrations in Snap Lake were 1.5 µg/L in 2012. Boron concentrations were elevated in all samples collected from Lake 13 in 2012. The maximum boron concentration was 9.8 µg/L in 2012 compared to 1.0 µg/L in 2005. As outlined in Appendix 3A, glass storage bottles likely contributed to the detectable concentrations of boron in the blank samples. Total manganese concentrations in 2012 were approximately double the concentrations measured in 2005 in Lake 13. The source of the increase in total manganese is unknown. However, for comparison, maximum total boron and manganese in Snap Lake in 2012 were 53 and 39 µg/L in Lake 13 in 2012 compared to 9.8 and 5.3 µg/L in Lake 13.

In summary, concentrations of most parameters in Lake 13 were consistent between 2005 and 2012. Exceptions were total boron, manganese and molybdenum. Follow-up sampling will determine whether these concentrations continue to be elevated, or were the result of potential isolated errors during the August sampling event. Regardless, concentrations of those three metals were much lower than measured in Snap Lake in 2012. Similarly, concentration of total dissolved solids, ions, nitrogen, nutrients, and the remaining metals were also low and similar to concentrations in Snap Lake prior to treated effluent discharge. Total phosphorus concentrations were comparable between lakes (further discussion on total phosphorus is provided in Section 3.4.4). On the basis of these data, Lake 13 is a suitable reference lake for the water quality component of the AEMP.

12.3.3.2 Sediment Quality

Lake 13 sediments had slightly lower total organic carbon than Snap Lake and Northeast Lake (Section 4.4.2.1). In Lake 13 sediments, concentrations of lead, mercury, and zinc were below their respective Interim Sediment Quality Guidelines (ISQG) at all five stations in 2012, and cadmium was above its ISQG at one station. Concentrations of chromium and copper were above their ISQGs at all five stations; this was consistent with observations for Snap Lake and Northeast Lake and suggests that concentrations are naturally elevated in sediments in the regional area surrounding the Mine.

Arsenic concentrations were higher in Lake 13 sediments than either Snap Lake or Northeast Lake, ranging from 5.0 to 37.2 milligram per kilogram dry weight (mg/kg dw); concentrations were above the ISQG at four stations and above the probable effects level at two 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 Environmental Assessment Report 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 within the range of natural variability of the region. It is recommended that sediment quality sampling in Lake 13 be repeated in 2013 to determine whether the elevated and variable concentrations of arsenic observed in 2012 are representative of actual conditions in Lake 13. However, despite the arsenic sediment data, on the basis of the overall sediment data collected in 2012, Lake 13 is a suitable reference lake for the sediment quality component of the AEMP.

12.3.3.3 Plankton

Chlorophyll

In August 2012, the chlorophyll *a* concentration collected at a single station in Lake 13 (0.68 µg/L) was lower than the mean August concentration in Snap Lake (0.99 ± 0.11 µg/L in the main basin and 1.1 ± 0.1 µg/L in the northwest arm), but similar to the mean August concentration in Northeast Lake (0.74 ± 0.03 µg/L; Figure 5-29). In contrast, the chlorophyll *c* concentration in Lake 13 (0.06 µg/L) was lower than the mean concentration in Snap Lake (0.09 ± 0.03 µg/L in the main basin and 0.07 ± 0.02 µg/L in the northwest arm) and Northeast Lake (0.11 ± 0.02 µg/L).

The 2012 concentrations of chlorophyll *a* and chlorophyll *c* in Lake 13 were similar to those collected in 2005 (0.95 µg/L and 0.16 µg/L respectively; Golder 2005).

Phytoplankton

Phytoplankton biomass in the main basin of Snap Lake (594 ± 79 milligrams per cubic metre [mg/m^3]) was greater than observed at the single station sampled in Lake 13 (443 mg/m^3). However, phytoplankton biomass in Lake 13 was slightly greater than in Northeast Lake (415 ± 59 mg/m^3) and the northwest arm of Snap Lake (405 ± 98 mg/m^3; Figure 5-29). Phytoplankton abundance in the main basin of Snap Lake was substantially greater than in Lake 13 (Appendix A5, Table 5). Phytoplankton abundance in the northwest arm of Snap Lake and Northeast Lake were less than in the main basin of Snap Lake, but greater than observed in Lake 13.

The proportion of major taxonomic groups of phytoplankton varied among all three lakes in August. In Lake 13, chrysophytes were the dominant phytoplankton group by biomass, followed by cyanobacteria and chlorophytes (Figure 5-30). Although a higher percentage of chrysophytes (48 percent [%]) and a lower percentage of cyanobacteria (25%) were observed in Lake 13 compared to Northeast Lake (15% chrysophytes and 70% cyanobacteria), the phytoplankton communities in these two lakes were most comparable. The non-metric multi-dimensional scaling (NMDS) results showed that stations in Snap Lake from 2004 and 2005 grouped closely with Northeast Lake (2011 and 2012) and Lake 13 (2012; Figure 5-31).

Zooplankton

Zooplankton abundance was higher in Lake 13 (58,147 organisms per cubic metre [org/m^3]) compared to the northwest arm of Snap Lake (30,780 ± 11,106 org/m^3), the main basin of Snap Lake (18,603 ± 1,638 org/m^3), and Northeast Lake (24,985 ± 3,228 org/m^3) (Figure 5-29). Similarly, zooplankton biomass was higher in Lake 13 (121,654 micrograms per cubic metre [$\mu g/m^3$]) compared to the northwest arm of Snap Lake (90,220 ± 32,900 $\mu g/m^3$), the main basin of Snap Lake (80,020 ± 7,728 $\mu g/m^3$), and Northeast Lake (65,020 ± 2,537 $\mu g/m^3$).

Zooplankton relative biomass in Lake 13 was unlike Snap Lake or Northeast Lake in August. Rotifers dominated the zooplankton community in Lake 13 (76%), while the zooplankton community in the northwest arm of Snap Lake was dominated by cyclopoid copepods (58%); the main basin and Northeast Lake zooplankton communities were dominated by calanoid copepods (49% and 42%, respectively; Figure 5-32).

Overall, on the basis of one sampling event in 2012, some aspects of the Lake 13 plankton community appear to be dissimilar from Snap Lake (pre-mining and during operations) and Northeast Lake. Additional data are required prior to making a definitive conclusion whether the Lake 13 plankton community is comparable to the other lakes and whether differences reflect natural variability in the region.

Benthic Invertebrates

In 2012, benthic invertebrate communities in Lake 13 were different than both Snap Lake and Northeast Lake. Total density, richness, *Microtendipes* density, Pisidiidae density, *Stictochironomus* density, *Valvata sincera* density and *Procladius* density were all higher in Lake 13 compared to Northeast Lake. Total organic carbon was also lower in Lake 13 compared to Northeast Lake and Snap Lake. On the basis of one year of data, the benthic invertebrate community of Lake 13 appears to be different from Snap Lake pre-mining. However, these differences may well represent natural variability in the region.

Fish Health

A total of five species of fish were captured in Lake 13 in 2012 during the small-bodied fish survey: Burbot, Lake Chub, Longnose Sucker, Ninespine Stickleback, and Slimy Sculpin (Table 7-6). The small-bodied species captured in 2012 are the same small-bodied species as captured in 2005, with the exception of the addition of Slimy Sculpin. Lake Trout, Round Whitefish, and Northern Pike were not captured in 2102 or in 2005. Fishing efforts and gear were focussed on small-bodied fish in 2012 rather than large-bodied fish as in 2005. Large-bodied fish will be sampled in Lake 13 in 2013 and the number and type of species will be reviewed at that time. Overall, the fish community of Lake 13 is similar to Snap Lake, with the exception of Northern Pike and Ninespine Stickleback which have been previously captured in Lake 13 but not Snap Lake. On the basis of the sampling to date, the number and type of fish species found in Lake 13 and Northeast Lake are identical.

The overall sizes of Lake Chub captured during the fish survey (length-frequency distribution) were not the same among Snap Lake, Northeast Lake, Lake 13, and Downstream Lake 1 (Section 7.4.2). This was possibly due to the different sampling methods used in each lake. Adult male and female and juvenile fish from Lake 13 were similar in length, weight, relative liver size, and gonad size (Section 7.4.3.3) to fish from Northeast Lake and were mostly similar to Snap Lake. Snap Lake and Downstream Lake 1 fish were shorter and lighter than the two reference lakes, while Northeast Lake fish were longer than fish from the other lakes. These apparent differences may be due to the different fishing methods used in different lakes. On the basis of one year of data, Lake 13 is an appropriate reference lake for Lake Chub fish health comparisons to Snap Lake and Northeast Lake.

Fish Tissue Chemistry

Lake Chub fish tissue was not collected during the 2005 reference lake survey; therefore, 2012 was the first time tissue chemistry data from Lake 13 were analyzed. Most parameters measured were statistically similar among the two reference lakes, Northeast Lake and Lake 13, and Snap Lake. Three parameters had significantly lower concentrations in Lake 13 Lake Chub relative to Northeast Lake: cadmium; cesium; and, sodium. Based on the above, the addition of a second reference lake helped distinguish natural variability from mine-related effect. On the basis of this one year of data, Lake 13 is an appropriate reference lake for Lake Chub tissue chemistry comparisons to Northeast Lake and Snap Lake.

12.3.4 Conclusion

Overall, water and sediment chemistry data collected in Lake 13 in 2012 were similar to data collected in 2005. There were differences in some biological components observed between the two reference lakes, Northeast Lake and Lake 13, and between Snap Lake pre-mining and Lake 13. However, the physical characteristics of the three lakes are comparable, and it is those characteristics that typically carry the heaviest weight during decisions regarding reference lake selection. Data from Lake 13 are expected to provide information on the range of natural variability within the region wherein Snap Lake is located.

12.3.5 Recommendations

Recommendations from the Suitability of Lake 13 as a Reference Lake Special Study are identified below.

- Lake 13 should be included in future AEMP sampling as a second reference lake for Snap Lake.
- Additional monitoring of the effects of the winter road (e.g., water, snowpack, dust) on Lake 13 should be included in the 2014 AEMP, as recommended by the MVLWB (MVLWB 2013).

12.3.6 References

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12.4 NUTRIENT STUDY

12.4.1 Introduction

12.4.1.1 Background

The 2012 Nutrient Special Study was designed as a follow-up study to the inter-laboratory comparison study conducted in 2011 (Golder 2011). The purpose of the 2011 inter-laboratory comparison study was to address data quality concerns related to phosphorus, and nitrogen data collected as part of the Aquatics Effects Monitoring Program (AEMP) for Snap Lake between 2008 and 2011 (Appendix 12D, Figures 12D-1 and 12D-2, respectively). A review of the nutrient data collected between 2008 and 2011 found inconsistencies with the nitrogen, and phosphorus results collected for the water quality and plankton components of the AEMP. Typically, the results of the nutrient samples from the plankton component were higher, and more variable than those from the water quality component. Nutrient samples collected for the water quality, and plankton components of the AEMP have been, and continue to be, submitted to the following three laboratories:

- ALS Canada Ltd. (ALS), the water quality component's primary laboratory;
- Maxxam Analytics Inc. (Maxxam), the water quality component quality control (QC) laboratory; and,
- the University of Alberta Biogeochemical Analytical Service Laboratory (UofA), the plankton component's laboratory.

These three laboratories were the focus of the 2011 comparison study, but due to uncertainty associated with the data of the 2011 inter-laboratory comparison study, additional inter-laboratory studies were required (Golder 2011).

12.4.1.2 Approach

The 2012 Nutrient Special Study was designed to further investigate inconsistencies observed in the nutrient data by answering the following three key questions:

1. Are the laboratories able to accurately measure known concentrations of nutrients?
2. Are there patterns in differences in the nutrient data provided by each laboratory?
3. How do nutrient concentrations in mid-depth grab samples compare to depth-integrated euphotic zone composite (herein referred to as euphotic zone) samples collected at the same station?

To answer the first key question, spike samples (i.e., samples of known concentrations) prepared by an International Standards Organization (ISO) 9001:2008 certified producer (Delta Scientific

Laboratory Products Ltd.) were sent to the three laboratories. The nutrient results for the three laboratories were compared to the known concentrations in the samples to assess the laboratories' ability to reliably analyze nutrients over a range of concentrations, particularly low-level nutrient concentrations that are routinely measured in the Snap Lake AEMP. To answer the second key question, the nutrient results from several split samples collected throughout the 2012 AEMP field program were reviewed to identify differences, and patterns in differences in results provided by the three laboratories. To answer the third key question, the results of samples collected at mid-depth, and in the euphotic zone from the three different laboratories were compared to identify differences in nutrient concentrations. The purpose of the third key question was to evaluate the potential for sampling depth to influence nutrient concentrations, and to provide guidance on future sampling practices for the water quality, and plankton components.

12.4.2 Laboratory Assessment

12.4.2.1 Key Question 1 - Are the laboratories able to accurately measure known concentrations of nutrients?

Spike Samples Preparation and Analysis

To evaluate each laboratory's ability to accurately measure known concentrations of various forms of nitrogen, and phosphorus, commercially purchased standard reference materials representing a range of concentrations from Delta Scientific Laboratory Products Ltd. were submitted to each laboratory as blind spike samples. Two separate series of spike samples were submitted in 2012: one series in July and one series in September (Table 12.4-1).

The July series of samples (SNP-N01 to SNP-N04) submitted to each laboratory included one blank sample, and three multi-parameter samples of varying concentrations of ammonia, nitrate, and phosphorus.

The September series of samples (SNP-N05 to SNP-N20) included a larger set of samples to further evaluate each laboratory. Similar to the July series, one blank sample, and three multi-parameter samples of varying concentrations of ammonia, nitrate, and phosphorus were submitted to each laboratory. In addition to these four sample types, 12 samples of varying concentrations, and sources of phosphorus were submitted for analysis.

The spike samples were made from analytical reagent (AR) grade reference standard sources of nitrogen, and phosphorus compounds, with 18 megaohm (MΩ) deionized water. The multi-parameter spike samples were prepared from ammonium chloride, sodium nitrate, and ammonium phosphate as the sources of ammonia, nitrate, and phosphorus, respectively. The spike samples in the September series that contained varying concentrations of phosphorus were prepared from two sources of phosphorus. Six samples were prepared from ammonium phosphate, and six were prepared from β-glycerophosphoric acid disodium salt pentahydrate.

B-glycerophosphoric acid disodium salt pentahydrate was used in an attempt to simulate phosphorus from an organic source, thereby investigating whether the laboratories' analytical methods were able to distinguish between ortho-phosphate, total dissolved, and total phosphorus.

The nitrogen, and phosphorus concentrations of the spike samples were based on the approximate range in concentrations observed in Northeast Lake, Snap Lake, and the treated effluent (Table 12.4-1). The low-level nutrient concentrations of spike samples SNP-N02, SNP-N06, SNP-N09, SNP-N12, SNP-N15, and SNP-N18 were consistent with approximate average concentrations in Northeast Lake. The mid-level nutrient concentrations of spike samples SNP-N03, SNP-N07, SNP-N10, SNP-N13, SNP-N16, and SNP-N19 were consistent with approximate average concentrations at the Snap Lake diffuser stations. The high-level concentrations of spike samples SNP-N04, SNP-N08, SNP-N11, SNP-N14, SNP-N17, and SNP-N20 were consistent with approximate concentrations in the treated effluent.

Table 12.4-1 Nutrient Concentrations in the 2012 Spike Samples

Spike ID	Ammonia (mg-N/L)	Nitrate (mg-N/L)	Total Nitrogen (mg-N/L)	Total Phosphorus (mg-P/L)
SNP-N01	0	0	0	0
SNP-N02	0.08	0.02	0.1	0.0015[(a)]
SNP-N03	0.7	2.8	3.5	0.004[(a)]
SNP-N04	2.5	10	12.5	0.01[(a)]
SNP-N05	0	0	0	0
SNP-N06	0.08	0.02	0.1	0.0015[(a)]
SNP-N07	0.7	2.8	3.5	0.004[(a)]
SNP-N08	2.5	10	12.5	0.01[(a)]
SNP-N09	0	0	0	0.0015[(a)]
SNP-N10	0	0	0	0.004[(a)]
SNP-N11	0	0	0	0.01[(a)]
SNP-N12	0	0	0	0.0015[(b)]
SNP-N13	0	0	0	0.004[(b)]
SNP-N14	0	0	0	0.01[(b)]
SNP-N15	0	0	0	0.0015[(a)]
SNP-N16	0	0	0	0.004[(a)]
SNP-N17	0	0	0	0.01[(a)]
SNP-N18	0	0	0	0.0015[(b)]
SNP-N19	0	0	0	0.004[(b)]
SNP-N20	0	0	0	0.01[(b)]

(a) The phosphorus source was ammonium phosphate.

(b) The phosphorus source was β-glycerophosphoric acid disodium salt pentahydrate.

mg-N/L = milligrams as nitrogen per litre; and mg-P/L = milligrams as phosphorus per litre; ID = identification number.

Each laboratory analyzed the spike samples for the following parameters: ammonia, nitrate, nitrate and nitrite, nitrite, total Kjeldahl nitrogen (TKN), total nitrogen (TN), dissolved inorganic phosphorus (DIP), dissolved organic phosphorus (DOP), ortho-phosphate, total dissolved phosphorus (TDP), total inorganic phosphorus (TIP), total organic phosphorus (TOP), and total phosphorus (TP). The analytical methods used by each laboratory are based on the reference methods presented in Table 12.4-2.

Table 12.4-2 Analytical Methods Used by Each Laboratory During the 2012 Nutrient Special Study

Parameter	ALS	Maxxam	UofA
Nitrogen			
Ammonia	SM 4500-NH_3	SM 4500-NH_3	SM 4500-NH3
Nitrate[(a)]	SM 4500-NO_3^-	SM 4500-NO_3^-	SM 4500-NO_3^-
Nitrate and nitrite	SM 4500-NO_2^- and SM 4500-NO_3^-	SM 4500-NO_2^- and SM 4500-NO_3^-	SM 4500-NO_2^- and SM 4500-NO_3^-
Nitrite	SM 4500-NO_2^-	SM 4500-NO_2^-	SM 4500-NO_2^-
Total Kjeldahl nitrogen	SM 4500-N_{org}	Calculation	Calculation
Total nitrogen	Calculation	SM-4500N	SM-4500N
Phosphorus			
Dissolved inorganic phosphorus	SM 4500-P	SM 4500-P E	SM 4500-P F
Dissolved organic phosphorus[(b)]	SM 4500-P	SM 4500-P E	SM 4500-P F
Ortho-Phosphate, as P	SM 4500-P	SM 4500-P E	SM 4500-P F
Total dissolved phosphorus	SM 4500-P	SM 4500-P E	SM 4500-P F
Total inorganic phosphorus	SM 4500-P	SM 4500-P E	SM 4500-P F
Total phosphorus	SM 4500-P	SM 4500-P E	SM 4500-P F
Total organic phosphorus[(c)]	SM 4500-P	SM 4500-P E	SM 4500-P F

(a) Nitrate was calculated by subtracting measured nitrite values from measured combined nitrate and nitrite values.

(b) Dissolved organic phosphorus was calculated by subtracting dissolved inorganic phosphorus values from dissolved phosphorus values.

(c) Total organic phosphorus was calculated by subtracting total inorganic phosphorus values from total phosphorus values.

ALS = ALS Canada Ltd.; Maxxam = Maxxam Analytics Inc.; UofA = University of Alberta Biogeochemical Analytical Service Laboratory; SM = Standard Methods for the Examination of Water and Wastewater (APHA 2012); NH_3 = ammonia; P = phosphorus; NO_3^- = nitrate; NO_2^- = nitrite.

All three laboratories based their analysis methods on the same analytical reference methods for all but two parameters: TKN and TN. The concentration of TKN is analyzed directly by ALS by Kjeldahl digestion at 380 degrees Celsius (°C) followed by automated colorimetric detection. Maxxam, and the UofA calculate TKN as follows:

$$(TKN) = (TN) – (NO_3^- \text{ and } NO_2^-) \quad \text{[Equation 12-2]}$$

Where:

(TKN) = calculated concentration of total Kjeldahl nitrogen;

(TN) = measured concentration of total nitrogen; and,

(NO_3^- and NO_2^-) = measured concentration of nitrate and nitrite.

Maxxam and the UofA analyze TN directly through persulfate oxidation of all digestible forms of nitrogen to nitrate followed by automated colorimetric detection. ALS did not provide TN results; TN results from ALS were calculated as follows:

$(TN) = (NO_3^- \text{ and } NO_2^-) + (TKN)$ [Equation 12-3]

Where:

(TN) = calculated concentration of total nitrogen;

(NO_3^- and NO_2^-) = measured concentration of nitrate and nitrite; and,

(TKN) = measured concentration of total Kjeldahl nitrogen.

Spike Samples Data Analysis

Each laboratory's ability to measure known concentrations of nutrients was evaluated based on the accuracy of the measured result, which was defined by the closeness of the measured value from each laboratory relative to the known concentration in the spike. Accuracy was assessed visually by plotting and comparing laboratory results to the known spike sample concentrations. Laboratory results were also compared to the known concentrations of the spike samples to determine the percent error, using the following equation:

$$\text{Percent Error} = \frac{\text{Laboratory Measured Concentration} - \text{Known Spike Concentration}}{\text{Known Spike Concentration}} \times 100$$ [Equation 12-4]

The absolute value of the percent error was compared to a minimum acceptance criterion of 20 percent (%). Absolute values of the percent errors of 20% or less were considered to have an acceptable level of accuracy.

In addition to evaluating accuracy, the results were also reviewed to identify possible patterns in biases in the data provided by each laboratory. Biases in the data might include consistently negative or positive percent error values outside the -20% to 20% range, which would indicate that the laboratory was consistently underestimating or overestimating, respectively, reported values.

Percent error calculations were completed for ammonia, nitrate, TKN, TN, ortho-phosphate, TDP, and TP. Laboratory results for the full suite of parameters listed in Table 12.4-2 are presented in Appendix 12D.

Percent error calculations were not completed for the two blank samples, SNP-N01, and SNP-N05. The blank samples were used to assess whether laboratories were reporting false positives, which was defined by the detection of nutrient concentrations when none were present.

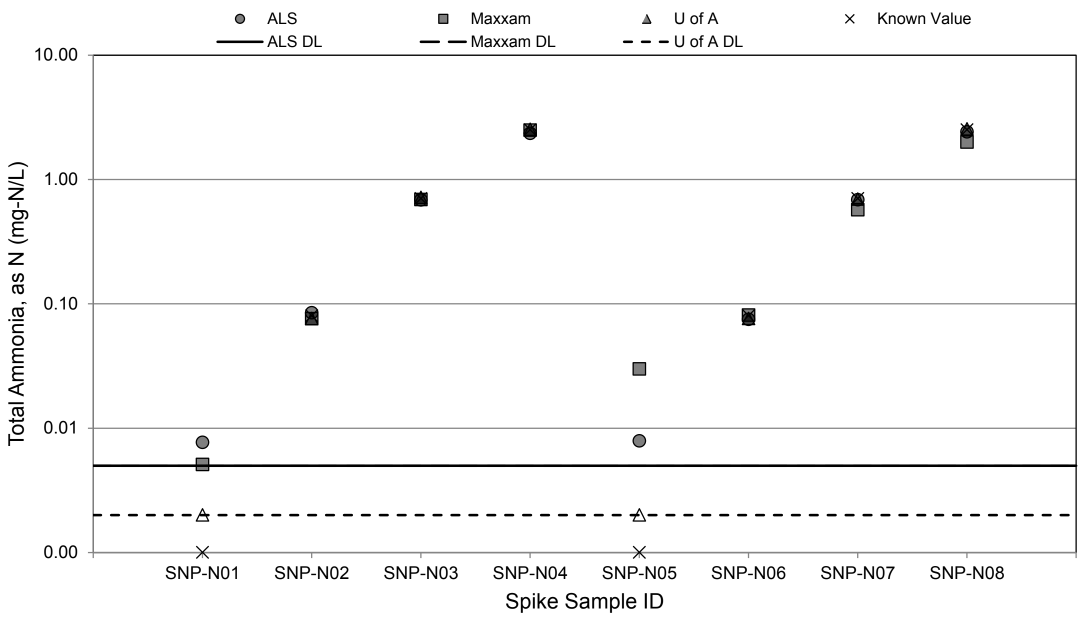
Spike Samples Results and Discussion

Nitrogen

All three laboratories reported a similar level of accuracy in the nitrogen analysis of the spike samples. Based on visual comparison of measured laboratory concentrations relative to known spike concentrations from the July, and September series of spike samples, measured laboratory concentrations were similar to the known spike concentrations for most of the samples, and for most of the nitrogen parameters (Figures 12.4-1 to 12.4-4).

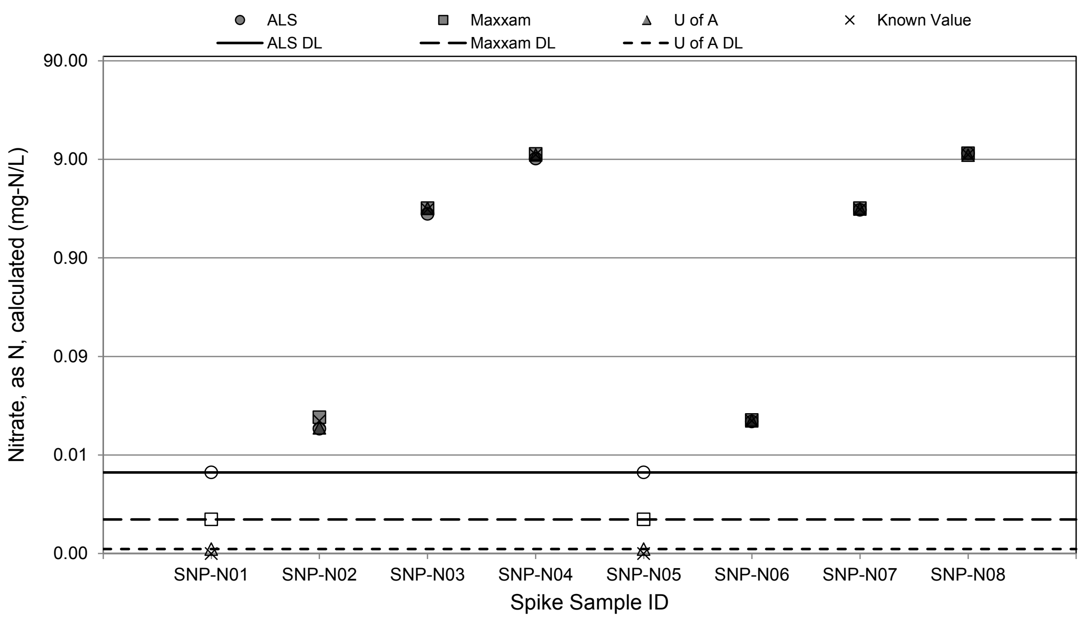
Ammonia and nitrate values were reported the most accurately by all three laboratories. The absolute percent error in ammonia and nitrate laboratory measurements met the acceptance criterion for all spike samples for all laboratories (Table 12.4-3).

Figure 12.4-1 Spike Samples Total Ammonia Laboratory Results



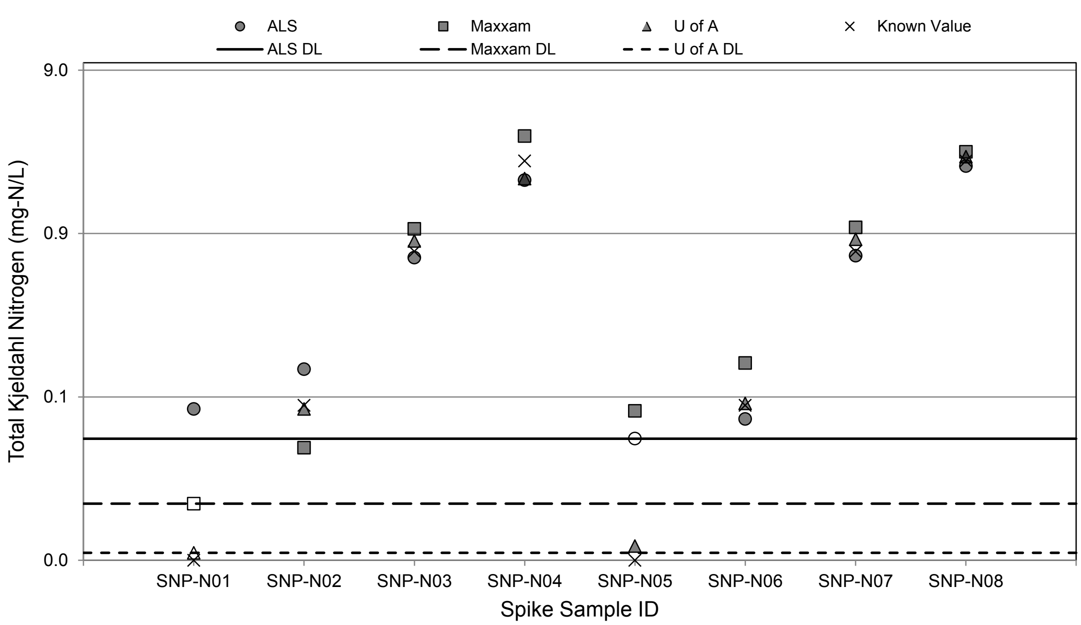
ALS = ALS Canada Ltd.; Maxxam = Maxxam Analytics Inc.; UofA = University of Alberta Biogeochemical Analytical Service Laboratory; DL = detection limit; mg = milligram; N = nitrogen; ID = identification number; L = litre; ALS DL = 0.005 mg-N/L; Maxxam DL = 0.005 mg-N/L; UofA DL = 0.002 mg-N/L; and open data points = laboratory results reported as less than detection limit.

Figure 12.4-2 Spike Samples Nitrate Laboratory Results



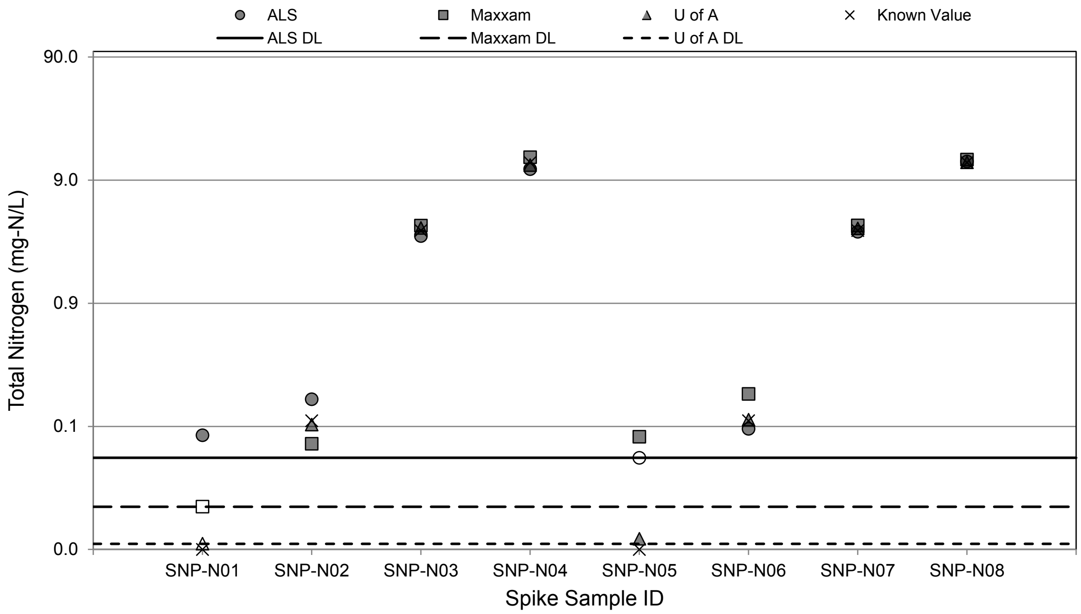
ALS = ALS Canada Ltd.; Maxxam = Maxxam Analytics Inc.; UofA = University of Alberta Biogeochemical Analytical Service Laboratory; DL = detection limit; mg = milligram; N = nitrogen; L = litre; ID = identification number; ALS DL = 0.006 mg-N/L; Maxxam DL = 0.002 mg-N/L; UofA DL = 0.001 mg-N/L; and open data points = laboratory results reported as less than detection limit.

Figure 12.4-3 Spike Samples Total Kjeldahl Nitrogen Laboratory Results



ALS = ALS Canada Ltd.; Maxxam = Maxxam Analytics Inc.; UofA = University of Alberta Biogeochemical Analytical Service Laboratory; DL = detection limit; mg-N/L = milligrams as nitrogen per litre; ; ID = identification number; ALS DL = 0.05 mg/L; Maxxam DL = 0.02 mg/L; UofA DL = 0.01 mg/L; and open data points = laboratory results reported as less than detection limit.

Figure 12.4-4 Spike Samples Total Nitrogen Laboratory Results



ALS = ALS Canada Ltd.; Maxxam = Maxxam Analytics Inc.; UofA = University of Alberta Biogeochemical Analytical Service Laboratory; DL = detection limit; mg-N/L = milligrams as nitrogen per litre; ID = identification number; ALS DL = 0.05 mg-N/L; Maxxam DL = 0.02 mg-N/L; UofA DL = 0.01 mg-N/L; and open data points = laboratory results reported as less than detection limit.

Table 12.4-3 Results of Nitrogen Spike Samples

Parameter	Known Spike Concentration (mg-N/L)	ALS			Maxxam			UofA		
		Detection Limit (mg-N/L)	Laboratory Measured Concentration (mg-N/L)	Percent Error (%)	Detection Limit (mg-N/L)	Laboratory Measured Concentration (mg-N/L)	Percent Error (%)	Detection Limit (mg-N/L)	Laboratory Measured Concentration (mg-N/L)	Percent Error (%)
SNP-N02										
Nitrate, as N, calculated[(a)]	0.02	0.006	0.017	-17	0.002	0.022	9	0.001[(b)]	0.017	-15
Total Ammonia, as N	0.08	0.005	0.085	6	0.005	0.076	-5	0.002	0.076	-5
Total Kjeldahl Nitrogen	0.08	0.05	0.13	**66**	0.02	0.04	**-45**	0.01	0.08	-5
Total Nitrogen[(c)]	0.1	0.05[(d)]	0.15	**50**	0.02	0.07	**-35**	0.01	0.09	-7
SNP-N03										
Nitrate, as N, calculated[(a)]	2.8	0.006	2.520	-10	0.01[(e)]	2.88	3	0.001[(b)]	2.860	2
Total Ammonia, as N	0.7	0.005	0.683	-2	0.005	0.689	-2	0.002	0.715	2
Total Kjeldahl Nitrogen	0.7	0.05	0.64[(f)]	-8	0.2[(e)]	1.0	**37**	0.01	0.81	16
Total Nitrogen[(c)]	3.5	0.05[(d)]	3.16	-10	0.2[(g)]	3.8	10	0.01	3.67	5
SNP-N04										
Nitrate, as N, calculated[(a)]	10	0.5[(h)]	9.2	-9	0.02[(e)]	10.20	2	0.001[(b)]	9.950	-1
Total Ammonia, as N	2.5	0.005	2.350	-6	0.05(e)	2.49	0	0.002	2.520	1
Total Kjeldahl Nitrogen	2.5	0.05	1.91[(f)]	**-24**	0.2[(e)]	3.6	**42**	0.01	1.95	**-22**
Total Nitrogen[(c)]	12.5	0.05[(d)]	11.06	-12	0.2[(e)]	13.8	10	0.01	11.90	-5
SNP-N06										
Nitrate, as N, calculated[(a)]	0.02	0.006	0.020	-2	0.002	0.020	2	0.001[(b)]	0.020	0
Total Ammonia, as N	0.08	0.005	0.075	-6	0.005	0.081	1	0.002	0.076	-5
Total Kjeldahl Nitrogen	0.08	0.05	0.07	-18	0.02	0.15	**81**	0.01	0.08	2
Total Nitrogen[(c)]	0.1	0.05[(d)]	0.09	-14	0.02	0.17	**65**	0.01	0.10	2
SNP-N07										
Nitrate, as N, calculated[(a)]	2.8	0.006	2.770	-1	0.004[(e)]	2.880	3	0.001[(b)]	2.820	1
Total Ammonia, as N	0.7	0.005	0.687	-2	0.005	0.570	-19	0.002	0.692	-1
Total Kjeldahl Nitrogen	0.7	0.05	0.66	-6	0.1[(e)]	1.0	**40**	0.01	0.83	19
Total Nitrogen[(c)]	3.5	0.05[(d)]	3.43	-2	0.1[(e)]	3.9	10	0.01	3.65	4

Table 12.4-3 Results of Nitrogen Spike Samples

Parameter	Known Spike Concentration (mg-N/L)	ALS			Maxxam			UofA		
		Detection Limit (mg-N/L)	Laboratory Measured Concentration (mg-N/L)	Percent Error (%)	Detection Limit (mg-N/L)	Laboratory Measured Concentration (mg-N/L)	Percent Error (%)	Detection Limit (mg-N/L)	Laboratory Measured Concentration (mg-N/L)	Percent Error (%)
SNP-N08										
Nitrate, as N, calculated[(a)]	10	0.5[(h)]	10.4	4	0.02[(e)]	10.40	4	0.001[(b)]	9.900	-1
Total Ammonia, as N	2.5	0.005	2.410	-4	0.025[(e)]	2.000	-20	0.002	2.540	2
Total Kjeldahl Nitrogen	2.5	0.05	2.33[(f)]	-7	0.2[(e)]	2.9	14	0.01	2.66	6
Total Nitrogen[(c)]	12.5	0.05[(d)]	12.73	2	0.2[(e)]	13.2	6	0.01	12.56	0.5

Note: Percent errors were calculated using data at the same precision level provided by the laboratory; tabulated laboratory data is rounded to the same significant digit as the highest detection limit associated with the sample. Boldface indicates absolute percent error value exceeds acceptance criterion of 20%. Gray shading indicates absolute percent value exceeds acceptance criterion of 20% and laboratory reported value is greater than five times the DL.

(a) Calculated UofA nitrate concentrations were equal to the combined measured nitrate and nitrite concentrations minus the measured nitrite concentrations.

(b) The DL for nitrate was equal to the DL for combined nitrate and nitrite values.

(c) ALS TN values were calculated as the sum of measured TKN and measured combined nitrate and nitrite values.

(d) The DL for TN was equal to the DL for TKN.

(e) The sample result from the original analysis was greater than the instrument calibration range; therefore the sample was diluted to bring the result within the instrument calibration range and reanalyzed, as a result, the DL was raised by the same factor used in the dilution.

(f) The laboratory result was flagged with the following qualifier: The TKN value was biased low, due to nitrate interference in the sample.

(g) The laboratory result was flagged with the following qualifier: The DL was raised due to sample matrix interference.

(h) The laboratory result was flagged with the following qualifier: The DL was adjusted for sample matrix effects.

ALS = ALS Canada Ltd.; Maxxam = Maxxam Analytics Inc.; UofA = University of Alberta Biogeochemical Analytical Service Laboratory; DL = detection limit; mg-N/L = milligrams as nitrogen per litre; % = percent; TKN = total Kjeldahl nitrogen; TN = total nitrogen.

Overall, nitrogen results from the UofA were more accurate than ALS and Maxxam; the absolute percent error in UofA laboratory measurements of all forms of nitrogen met the acceptance criterion for all but one TKN result (Table 12.4-3). The average absolute percent error for all nitrogen results from the UofA was 5% (Table 12.4-4). For ALS, the absolute percent error in TN, and TKN laboratory measurements was within the acceptance criterion in all but one and two samples, respectively.

The absolute percent error in TKN results reported by Maxxam did not meet the acceptance criterion in five of the six samples, of which four were above the known concentration (Tables 12.4-3 and 12.4-4); this suggests that TKN results from Maxxam may be biased high. Reported values of TKN from Maxxam are directly related to TN (Equation 1 in Section 12.4.2.1 Spike Samples Preparation and Analysis), and although the absolute percent errors of all but two TN results from Maxxam were within the acceptance criterion, five of the six TN results were above the known sample concentrations. Because concentrations of TKN are lower than TN, the error makes up a larger portion of the reported TKN value. While Maxxam's overestimations in TN measurements still met the acceptance criterion in four of the six cases, the overestimation was magnified in their calculated TKN values because of the relatively lower concentrations reported. Three of the five TKN results reported by ALS were flagged as biased low due to high nitrate interference. This interference is not considered to be an issue in the AEMP, as the concentrations of nitrate in these samples were approximately five times higher than the concentrations measured in Snap Lake.

Results from the blank sample assessment demonstrated that the UofA reported the most accurate results, and Maxxam reported the least accurate results (Table 12.4-5). The UofA, which has the lowest detection limits, did not report any values above their detection limits, and reported only two parameters, TKN and TN, equal to the respective detection limits. Maxxam and ALS each reported four values above their detection limits in the blank samples. All four ALS results would not pass ALS' internal quality assurance/quality control (QA/QC) criterion for a blank sample, which was defined as any value equal to or greater than the detection limit (ALS 2012). Maxxam has less stringent QA/QC criterion for blank samples than ALS; the Maxxam criterion is defined as less than two times the detection limit (Maxxam 2012). All four ALS results would pass Maxxam's QA/QC criterion for a blank sample; however, the detectable TKN, and TN results reported by Maxxam would not pass their internal QA/QC criterion of a blank sample.

In summary, all three laboratories provided accurate nitrogen data, with the exception of the TKN results provided by Maxxam that suggest a possible high bias. Overall, the UofA provided the most accurate nitrogen data followed by ALS, then Maxxam.

Table 12.4-4 Summary of Nitrogen Spike Samples

Parameter	Results Outside Acceptance Criterion				All Results			
	Low	High	Total	Average Absolute Percent Error (%)	Below Known Spike Concentration	Above Known Spike Concentration	Equal to Known Spike Concentration	Average Absolute Percent Error (%)
Nitrate, as N, calculated								
ALS	0	0	**0**	N/A	5	1	0	7
Maxxam	0	0	**0**	N/A	0	6	0	4
UofA	0	0	**0**	N/A	3	2	1	3
Total Ammonia, as N								
ALS	0	0	**0**	N/A	5	1	0	4
Maxxam	0	0	**0**	N/A	4	1	1	8
UofA	0	0	**0**	N/A	3	3	0	3
Total Kjeldahl Nitrogen								
ALS	1	1	**2**	45	5	1	0	21
Maxxam	1	4	**5**	49	1	5	0	43
UofA	1	0	**1**	22	2	4	0	12
Total Nitrogen								
ALS	0	1	**1**	50	4	2	0	15
Maxxam	1	1	**2**	50	1	5	0	23
UofA	0	0	**0**	N/A	2	4	0	4
All Nitrogen Results								
ALS	1	2	**3**	47	19	5	0	12
Maxxam	2	5	**7**	50	6	17	1	19
UofA	1	0	**1**	22	10	13	1	5

Note: Low = number of sample results with percent error less than -20%; High = number of sample results with percent error greater than 20%;
% = percent; N/A = not applicable because no results exceed acceptable criterion; ALS = ALS Canada Ltd.; Maxxam = Maxxam Analytics Inc.; UofA = University of Alberta Biogeochemical Analytical Service Laboratory.

Table 12.4-5 Nitrogen Results from Blank Samples

Parameter	Known Spike Concentration (mg-N/L)	ALS		Maxxam		UofA	
		Detection Limit	Laboratory Measured Concentration (mg-N/L)	Detection Limit	Laboratory Measured Concentration (mg-N/L)	Detection Limit	Laboratory Measured Concentration (mg-N/L)
SNP-N01							
Nitrate, as N, calculated[(a)]	0	0.006	<0.006	0.002	<0.002	0.001[(b)]	<0.001
Total Ammonia, as N	0	0.005	**0.008**	0.005	**0.005**	0.002	<0.002
Total Kjeldahl Nitrogen	0	0.05	**0.08**	0.02	<0.02	0.01	<0.01
Total Nitrogen[(c)]	0	0.05[(d)]	**0.08**	0.02	<0.02	0.01	<0.01
SNP-N05							
Nitrate, as N, calculated[(a)]	0	0.006	<0.006	0.002	<0.002	0.001[(b)]	<0.001
Total Ammonia, as N	0	0.005	**0.008**	0.005	**0.030**	0.002	<0.002
Total Kjeldahl Nitrogen	0	0.05	<0.05	0.02	**0.07**	0.01	**0.01**
Total Nitrogen[(c)]	0	0.05[(d)]	<0.05	0.02	**0.07**	0.01	**0.01**

Note: **Bold** values reported at or above the detection limit.

(a) Calculated UofA nitrate concentrations were equal to the combined measured nitrate and nitrite concentrations minus the measured nitrite concentrations.

(b) The DL for nitrate was equal to the DL for combined nitrate and nitrite values.

(c) ALS TN values were calculated as the sum of measured TKN and measured combined nitrate and nitrite values.

(d) The DL for TN was equal to the DL for TKN.

ALS = ALS Canada Ltd.; Maxxam = Maxxam Analytics Inc.; UofA = University of Alberta Biogeochemical Analytical Service Laboratory; mg-N/L = milligrams as nitrogen per litre; <= less than; TKN = total Kjeldahl nitrogen; TN = total nitrogen.

Phosphorus

The three laboratories displayed a lower level of accuracy in the analysis of the phosphorus spike samples relative to the performance of the three laboratories in the analysis of the nitrogen spike samples (Figures 12.4-5 to 12.4-7).

The absolute percent error in TDP laboratory measurements did not meet the acceptable criterion for accuracy in approximately one quarter to one third of the spike samples, depending on the laboratory (Table 12.4-6). The lowest rate of error occurred in the ALS results where 4 out of 18 samples did not meet the criterion. However, the TDP results from ALS may be biased low because the majority of ALS laboratory results were lower than the known spike concentrations. The highest rate of error occurred in the UofA results, where 7 out of the 18 samples did not meet the criterion.

The absolute percent error in ortho-phosphate laboratory measurements did not meet the acceptable criterion in approximately half of the spike samples for all laboratories (Table 12.4-6). The lowest rate of error occurred in the Maxxam results, where 8 of the 18 laboratory results did not meet the criterion. Both UofA, and ALS appeared to have a low bias for ortho-phosphate results because laboratory measurements for UofA, and ALS were below the known spike concentrations in almost all samples (Table 12.4-7).

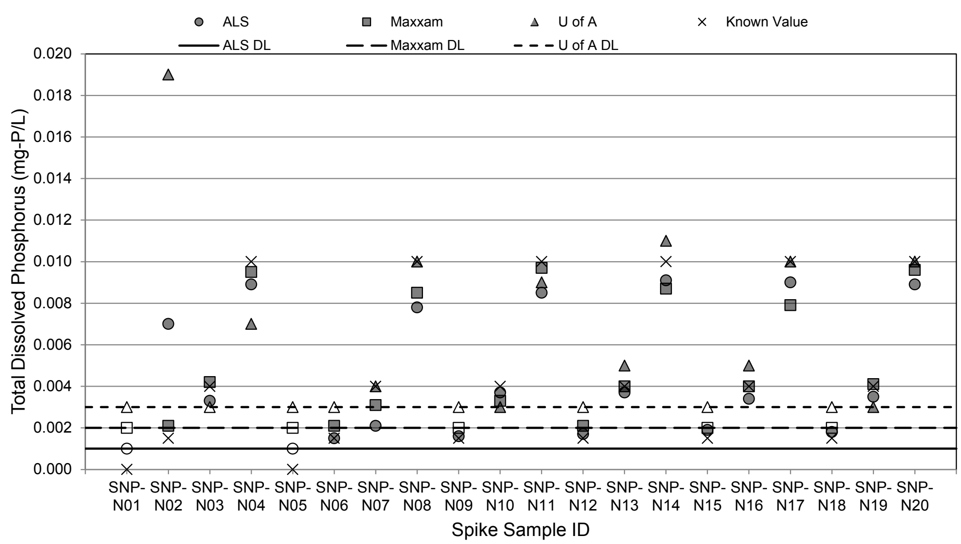
The absolute percent error in TP laboratory measurements did not meet the acceptable criterion in approximately one quarter to one third of the spike samples, depending on the laboratory (Table 12.4-6). No consistent pattern in any one laboratory overestimating or underestimating TP concentrations was observed.

Based on all of the spike samples submitted, Maxxam reported the fewest results outside the acceptance criterion and had the lowest average absolute percent error for all forms of phosphorus: 20% compared to 37% for ALS and 96% for the UofA (Table 12.4-7). Notably high concentration results from ALS and UofA for one of the spike samples (SNP-N02) contributed a heavy weighting to their average absolute percent error because UofA, and to a lesser degree ALS, grossly over-estimated the amount of phosphorus in the sample. If this spike sample is removed from the assessment, the average absolute percent error for all of the phosphorus results dropped from 96% to 14% for the UofA, 37% to 20% for ALS, and 20% to 19% for Maxxam (Tables 12.4-7 and 12.4-8).

The amount of phosphorus in the spike samples influenced the accuracy of all three laboratories to measure phosphorus. All three laboratories reported the fewest results outside the acceptance criteria in the spike samples with the higher phosphorus concentration of 0.01 mg-P/L (SNP-N04, SNP-N08, SNP-N1, SNP-N14, SNP-N17, and SNP-N20). Approximately 90% of the results met the acceptance criterion for accuracy when the spike concentration was 0.01 mg-P/L (Table 12.4-6). The laboratories had the lowest accuracy when analyzing the spike samples with the lower phosphorus concentration of 0.0015 mg-P/L (SNP-N02, SNP-N06, SNP-N09, SNP-N12, SNP-N15, and SNP-N18). Only about 10% of the results for these lower level spike samples were within the acceptable criterion for accuracy. This is not surprising given that this method is designed to be most reliable at 10 to 200 micrograms per litre (µg/L), not 2 to 3 µg/L (APHA 2012).

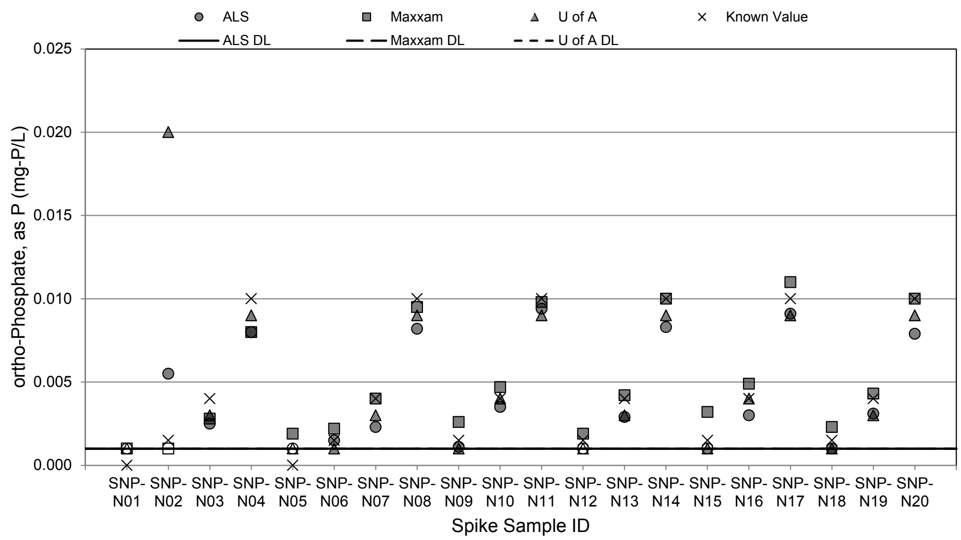
In the blank sample assessment, the laboratories reported phosphorus results either below the detection limit or within two times the detection limit (Table 12.4-9). Blank results from UofA were all below the detection limit. Maxxam and ALS reported two results above the detection limits; however, both results were within two times the detection limit and would have passed Maxxam's internal QA/QC criterion for a blank sample (Maxxam 2012).

Figure 12.4-5 Spike Samples Total Dissolved Phosphorus Laboratory Results



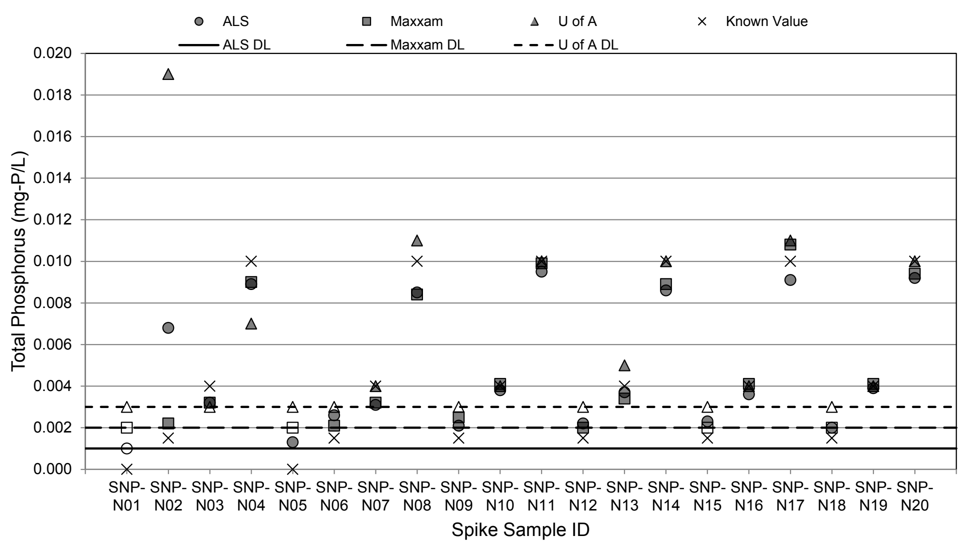
ALS = ALS Canada Ltd.; Maxxam = Maxxam Analytics Inc.; UofA = University of Alberta Biogeochemical Analytical Service Laboratory; DL = detection limit; mg-P/L = milligrams as phosphorus per litre; ID = identification number; ALS DL = 0.001 mg-P/L; Maxxam DL = 0.002 mg-P/L; UofA DL = 0.003 mg-P/L; and open data points = laboratory results reported as less than detection limit.

Figure 12.4-6 Spike Samples ortho-Phosphate Laboratory Results



ALS = ALS Canada Ltd.; Maxxam = Maxxam Analytics Inc.; UofA = University of Alberta Biogeochemical Analytical Service Laboratory; DL = detection limit; mg-P/L = milligrams as phosphorus per litre; ID = identification number; ALS DL = 0.001 mg-P/L; Maxxam DL = 0.001 mg-P/L; UofA DL = 0.001 mg-P/L; and open data points = laboratory results reported as less than detection limit.

Figure 12.4-7 Spike Samples Total Phosphorus Laboratory Results



ALS = ALS Canada Ltd.; Maxxam = Maxxam Analytics Inc.; UofA = University of Alberta Biogeochemical Analytical Service Laboratory; DL = detection limit; mg-P/L = milligrams as phosphorus per litre; ID = identification number; ALS DL = 0.001 mg-P/L; Maxxam DL = 0.002 mg-P/L; UofA DL = 0.003 mg-P/L; and open data points = laboratory results reported as less than detection limit.

Table 12.4-6 Results of Phosphorus Spike Samples

Parameter	Known Spike Concentration (mg-P/L)	ALS			Maxxam			UofA		
		Detection Limit (mg-P/L)	Laboratory Measured Concentration (mg-P/L)	Percent Error (%)[a]	Detection Limit (mg-P/L)	Laboratory Measured Concentration (mg-P/L)	Percent Error (%)[a]	Detection Limit (mg-P/L)	Laboratory Measured Concentration (mg-P/L)	Percent Error (%)[a]
SNP-N02										
ortho-Phosphate, as P	0.0015	0.001	0.006	**267**	0.001	<0.001	**-33**	0.001	0.020	**1233**
Total Dissolved Phosphorus	0.0015	0.001	0.007	**367**	0.002	0.002	***40***	0.003	0.019	***1167***
Total Phosphorus	0.0015	0.001	0.007	**353**	0.002	0.002	***47***	0.003	0.019	***1167***
SNP-N03										
ortho-Phosphate, as P	0.004	0.001	0.003	**-38**	0.001	0.003	**-30**	0.001	0.003	**-25**
Total Dissolved Phosphorus	0.004	0.001	0.003	-18	0.002	0.004	5	0.003	<0.003	**-25**
Total Phosphorus	0.004	0.001	0.003	-20	0.002	0.003	-20	0.003	<0.003	**-25**
SNP-N04										
ortho-Phosphate, as P	0.01	0.001	0.008	-20	0.001	0.008	-20	0.001	0.009	-10
Total Dissolved Phosphorus	0.01	0.001	0.009	-11	0.002	0.010	-5	0.003	0.007	**-30**
Total Phosphorus	0.01	0.001	0.009	-11	0.002	0.009	-10	0.003	0.007	**-30**
SNP-N06										
ortho-Phosphate, as P	0.0015	0.001	0.002	0	0.001	0.002	**47**	0.001	0.001	**-33**
Total Dissolved Phosphorus	0.0015	0.001	0.002	0	0.002	0.002	***40***	0.003	<0.003	NC
Total Phosphorus	0.0015	0.001	0.003	**73**	0.002	0.002	***40***	0.003	<0.003	NC
SNP-N07										
ortho-Phosphate, as P	0.004	0.001	0.002	**-43**	0.001	0.004	0	0.001	0.003	**-25**
Total Dissolved Phosphorus	0.004	0.001	0.002	**-48**	0.002	0.003	**-23**	0.003	0.004	0
Total Phosphorus	0.004	0.001	0.003	**-23**	0.002	0.003	-20	0.003	0.004	0
SNP-N08										
ortho-Phosphate, as P	0.01	0.001	0.008	-18	0.001	0.010	-5	0.001	0.009	-10
Total Dissolved Phosphorus	0.01	0.001	0.008	**-22**	0.002	0.009	-15	0.003	0.010	0
Total Phosphorus	0.01	0.001	0.009	-15	0.002	0.008	-16	0.003	0.011	10
SNP-N09										
ortho-Phosphate, as P	0.0015	0.001	0.001	**-27**	0.001	0.003	**73**	0.001	0.001	**-33**
Total Dissolved Phosphorus	0.0015	0.001	0.002	7	0.002	<0.002	NC	0.003	<0.003	NC
Total Phosphorus	0.0015	0.001	0.002	**40**	0.002	0.003	***67***	0.003	<0.003	NC
SNP-N10										
ortho-Phosphate, as P	0.004	0.001	0.004	-13	0.001	0.005	18	0.001	0.004	0
Total Dissolved Phosphorus	0.004	0.001	0.004	-8	0.002	0.003	-18	0.003	0.003	**-25**
Total Phosphorus	0.004	0.001	0.004	-5	0.002	0.004	3	0.003	0.004	0

Table 12.4-6 Results of Phosphorus Spike Samples

Parameter	Known Spike Concentration (mg-P/L)	ALS			Maxxam			UofA		
		Detection Limit (mg-P/L)	Laboratory Measured Concentration (mg-P/L)	Percent Error (%)[(a)]	Detection Limit (mg-P/L)	Laboratory Measured Concentration (mg-P/L)	Percent Error (%)[(a)]	Detection Limit (mg-P/L)	Laboratory Measured Concentration (mg-P/L)	Percent Error (%)[(a)]
SNP-N11										
ortho-Phosphate, as P	0.01	0.001	0.009	-6	0.001	0.010	-2	0.001	0.009	-10
Total Dissolved Phosphorus	0.01	0.001	0.009	-15	0.002	0.010	-3	0.003	0.009	-10
Total Phosphorus	0.01	0.001	0.010	-5	0.002	0.010	-1	0.003	0.010	0
SNP-N12										
ortho-Phosphate, as P	0.0015	0.001	<0.001	**-33**	0.001	0.002	**27**	0.001	<0.001	**-33**
Total Dissolved Phosphorus	0.0015	0.001	0.002	13	0.002	0.002	***40***	0.003	<0.003	NC
Total Phosphorus	0.0015	0.001	0.002	**47**	0.002	0.002	***33***	0.003	<0.003	NC
SNP-N13										
ortho-Phosphate, as P	0.004	0.001	0.003	**-28**	0.001	0.004	5	0.001	0.003	**-25**
Total Dissolved Phosphorus	0.004	0.001	0.004	-8	0.002	0.004	0	0.003	0.005	**25**
Total Phosphorus	0.004	0.001	0.004	-8	0.002	0.003	-15	0.003	0.005	**25**
SNP-N14										
ortho-Phosphate, as P	0.01	0.001	0.008	-17	0.001	0.010	0	0.001	0.009	-10
Total Dissolved Phosphorus	0.01	0.001	0.009	-9	0.002	0.009	-13	0.003	0.011	10
Total Phosphorus	0.01	0.001	0.009	-14	0.002	0.009	-11	0.003	0.010	0
SNP-N15										
ortho-Phosphate, as P	0.0015	0.001	<0.001	**-33**	0.001	0.003	**113**	0.001	0.001	**-33**
Total Dissolved Phosphorus	0.0015	0.001	0.002	**27**	0.002	<0.002	NC	0.003	<0.003	NC
Total Phosphorus	0.0015	0.001	0.002	**53**	0.002	<0.002	NC	0.003	<0.003	NC
SNP-N16										
ortho-Phosphate, as P	0.004	0.001	0.003	**-25**	0.001	0.005	**23**	0.001	0.004	0
Total Dissolved Phosphorus	0.004	0.001	0.003	-15	0.002	0.004	0	0.003	0.005	**25**
Total Phosphorus	0.004	0.001	0.004	-10	0.002	0.004	3	0.003	0.004	0
SNP-N17										
ortho-Phosphate, as P	0.01	0.001	0.009	-9	0.001	0.011	10	0.001	0.009	-10
Total Dissolved Phosphorus	0.01	0.001	0.009	-10	0.002	0.008	**-21**	0.003	0.010	0
Total Phosphorus	0.01	0.001	0.009	-9	0.002	0.011	8	0.003	0.011	10
SNP-N18										
ortho-Phosphate, as P	0.0015	0.001	0.001	**-33**	0.001	0.002	**53**	0.001	0.001	**-33**
Total Dissolved Phosphorus	0.0015	0.001	0.002	20	0.002	<0.002	NC	0.003	<0.003	NC
Total Phosphorus	0.0015	0.001	0.002	**33**	0.002	<0.002	NC	0.003	<0.003	NC

Table 12.4-6 Results of Phosphorus Spike Samples

Parameter	Known Spike Concentration (mg-P/L)	ALS			Maxxam			UofA		
		Detection Limit (mg-P/L)	Laboratory Measured Concentration (mg-P/L)	Percent Error (%)[(a)]	Detection Limit (mg-P/L)	Laboratory Measured Concentration (mg-P/L)	Percent Error (%)[(a)]	Detection Limit (mg-P/L)	Laboratory Measured Concentration (mg-P/L)	Percent Error (%)[(a)]
SNP-N19										
ortho-Phosphate, as P	0.004	0.001	0.003	**-23**	0.001	0.004	8	0.001	0.003	**-25**
Total Dissolved Phosphorus	0.004	0.001	0.004	-13	0.002	0.004	3	0.003	0.003	**-25**
Total Phosphorus	0.004	0.001	0.004	-3	0.002	0.004	3	0.003	0.004	0
SNP-N20										
ortho-Phosphate, as P	0.01	0.001	0.008	**-21**	0.001	0.010	0	0.001	0.009	-10
Total Dissolved Phosphorus	0.01	0.001	0.009	-11	0.002	0.010	-4	0.003	0.010	0
Total Phosphorus	0.01	0.001	0.009	-8	0.002	0.009	-6	0.003	0.010	0

Note: Percent errors were calculated using data at the same precision level provided by the laboratory; tabulated laboratory data is rounded to the same significant digit as the highest detection limit associated with the sample. Boldface indicates absolute percent error value exceeds acceptance criterion of 20%. Gray shading indicates absolute percent value exceeds acceptance criterion of 20% and laboratory reported value is greater than five times the DL. Italics indicate a laboratory reported value where the known spike concentration is less than the DL.

(a) If the laboratory value was reported as less than the detection limit, the respective laboratory's detection limit was used for calculation purposes.

ALS = ALS Canada Ltd.; Maxxam = Maxxam Analytics Inc.; UofA = University of Alberta Biogeochemical Analytical Service Laboratory; mg-P/L = milligrams as phosphorus per litre; <= less than; NC = non-calculable (percent error NC because known spike concentration is less than DL); DL = detection limit.

Table 12.4-7 Summary of Phosphorus Spike Samples

Parameter	Results Outside Acceptance Criterion				All Results				
	Low	High	Total	Average Absolute Percent Error (%)	Below Known Spike Concentration	Above Known Spike Concentration	Non-Calculable	Equal to Known Spike Concentration	Average Absolute Percent Error (%)
ortho-Phosphate, as P									
ALS	10	1	**11**	52	16	1	0	1	36
Maxxam	2	6	**8**	50	5	10	0	3	26
UofA	9	1	**10**	150	15	1	0	2	87
Total Dissolved Phosphorus									
ALS	3	1	**4**	116	12	5	0	1	34
Maxxam	2	3	**5**	33	8	5	3	2	15
UofA	4	3	**7**	189	5	4	5	4	103
Total Phosphorus									
ALS	1	6	**7**	89	12	6	0	0	41
Maxxam	0	4	**4**	47	8	8	2	0	19
UofA	2	2	**4**	312	2	4	5	7	97
All Phosphorus Results									
ALS	14	8	**22**	85	40	12	0	2	37
Maxxam	4	13	**17**	43	21	23	5	5	20
UofA	15	6	**21**	217	22	9	10	13	96

Note: Low = number of sample results with percent error less than -20%; High = number of sample results with percent error greater than 20%.

% = percent; Not calculable = percent error non-calculable because known spike concentration was less than DL; ALS = ALS Canada Ltd.; Maxxam = Maxxam Analytics Inc.; UofA = University of Alberta Biogeochemical Analytical Service Laboratory.

Table 12.4-8 Summary of Phosphorus Spike Samples with Spike Sample SNP-N02 Removed

Parameter	Results Outside Acceptance Criterion				All Results				
	Low	High	Total	Average Absolute Percent Error (%)	Below Known Spike Concentration	Above Known Spike Concentration	Non-Calculable	Equal to Known Spike Concentration	Average Absolute Percent Error (%)
ortho-Phosphate, as P									
ALS	10	0	**10**	30	16	0	0	1	23
Maxxam	1	6	**7**	52	4	10	0	3	25
UofA	9	0	**9**	30	15	0	0	2	19
Total Dissolved Phosphorus									
ALS	3	0	**3**	32	12	4	0	1	15
Maxxam	2	2	**4**	31	8	4	3	2	13
UofA	4	2	**6**	26	5	3	5	4	15
Total Phosphorus									
ALS	1	5	**6**	45	12	5	0	0	22
Maxxam	0	3	**3**	47	8	7	2	0	17
UofA	2	1	**3**	27	2	3	5	7	8
All Phosphorus Results									
ALS	14	5	**19**	41	40	9	0	2	20
Maxxam	3	11	**14**	43	20	21	5	5	19
UofA	15	3	**18**	28	22	6	10	13	14

Note: Low = number of sample results with percent error less than -20%; High = number of sample results with percent error greater than 20%.

% = percent; Non-calculable = percent error non-calculable because known spike concentration was less than DL; ALS = ALS Canada Ltd.; Maxxam = Maxxam Analytics Inc.; UofA = University of Alberta Biogeochemical Analytical Service Laboratory.

Table 12.4-9 Phosphorus Results from Blank Samples

Parameter	Known Spike Concentration (mg-P/L)	ALS		Maxxam		UofA	
		Detection Limit	Laboratory Measured Concentration (mg-P/L)	Detection Limit	Result	Detection Limit	Laboratory Measured Concentration (mg-P/L)
SNP-N01							
ortho-Phosphate, as P	0	0.001	<0.001	0.001	<0.001	0.001	<0.001
Total Dissolved Phosphorus	0	0.001	<0.001	0.002	<0.002	0.003	<0.003
Total Phosphorus	0	0.001	<0.001	0.002	<0.002	0.003	<0.003
SNP-N05							
ortho-Phosphate, as P	0	0.001	<0.001	0.001	**0.002**	0.001	<0.001
Total Dissolved Phosphorus	0	0.001	<0.001	0.002	<0.002	0.003	<0.003
Total Phosphorus	0	0.001	**0.001**	0.002	<0.002	0.003	<0.003

Note: **Bold** values reported at or above the detection limit.

ALS = ALS Canada Ltd.; Maxxam = Maxxam Analytics; UofA = University of Alberta Biogeochemical Analytical Service Laboratory; mg-P/L = milligrams as phosphorus per litre; < = less than.

12.4.2.2 Key Question 2 - Are there patterns in the differences in the nutrient data provided by each laboratory?

Split Sample Collection and Analysis

The results from the 25 split samples collected throughout the 2012 AEMP field program were used to identify patterns in the differences in the nutrient data provided by the three laboratories. Split samples were collected, and sent to the three laboratories during the ice-covered season (February, March, April, and May 2012), and during the open-water season (July, August, and September 2012). The split samples used in the assessment are listed in Table 12.4-10.

Table 12.4-10 2012 Aquatic Effects Monitoring Program Split Samples used in Nutrient Data Assessment

Station	Location Description	Collection Date	Sample Control Number
NEL01	Northeast Lake	2012-02-21	2012-5026
NEL03		2012-07-10	2012-5178
NEL04		2012-07-10	2012-5179
NEL05		2012-02-21	2012-5030
SNAP02A	northwest arm of Snap Lake	2012-09-12	2012-5251/7001
SNAP08	far-field area near outlet of Snap Lake	2012-02-17	2012-5019
		2012-07-13	2012-5197/5202/5204
		2012-09-11	2012-5243/7005
SNAP11A	mid-field area of Snap Lake	2012-02-17	2012-5022
		2012-09-13	2012-5246/7002
SNP 02-20d	diffuser stations in the near-field area of Snap Lake	2012-03-18	2012-5066
		2012-07-09	2012-5155
		2012-08-12	2012-5207
		2012-12-14	2012-7031
SNP 02-20e		2012-02-19	2012-5038/42
		2012-03-18	2012-5062
		2012-07-09	2012-5158
		2012-08-12	2012-5210
		2012-09-14	2012-7033
SNP 02-20f		2012-02-19	2012-5047/48
		2012-03-21	2012-5069
		2012-05-13	2012-5131/34
		2012-07-09	2012-5161
		2012-08-12	2012-5213
		2012-09-14	2012-7032

Notes: date format is YYYY-MM-DD, where Y= year, M = month and D = day.

The split samples were collected according to standard water quality sampling methods (Environment Canada 1983, 2012). Water from each station was collected at mid-depth using a polyvinyl chloride (PVC) Kemmerer sampler. During the open-water season, the water collected at each station was poured from the Kemmerer sampler directly into three sets of appropriate sampling bottles, one set for each laboratory, unless a sample required filtration (for dissolved nutrient analyses). Samples collected for dissolved nutrients were poured from the Kemmerer into a clean 1-litre (L) laboratory-grade sampling container and filtered when the field crew returned to the De Beers Canada Inc. (De Beers) water processing facility at the end of the sampling day.

During the ice-covered season, a gasoline-powered ice auger was used to drill a hole through the ice so that the Kemmerer sampler could be lowered through the hole into the water column to collect the water samples. During the ice-covered programs, water collected at each station using the Kemmerer sampler was poured directly into 1 L laboratory grade sampling containers instead of the 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 filled, and filtered as required, from the 1 L containers when the field crew returned to the De Beers water processing facility at the end of the sampling day.

The split samples were analyzed for the same parameters, using the same methods described in Section 12.4.2.1 Spike Samples Preparation and Analysis.

Split Samples Data Analysis

The data analysis focused on ammonia, nitrate, ortho-phosphate, TDP, TKN, TN, and TP. Results from the three laboratories were plotted for each parameter, and visually reviewed to determine whether there were patterns in the differences in the data sets provided by each laboratory.

Split Samples Results and Discussion

Results from split samples collected throughout the 2012 AEMP field program were used to determine whether there were consistent patterns in any differences in the nutrient data provided by each laboratory.

Nitrogen

No consistent pattern in differences between laboratory results for ammonia or total nitrogen split samples were observed (Figures 12.4-8 and 12.4-9, respectively).

Nitrate results provided by ALS and Maxxam were generally similar to each other. However, nitrate results provided by the UofA were consistently lower than those provided by ALS, and

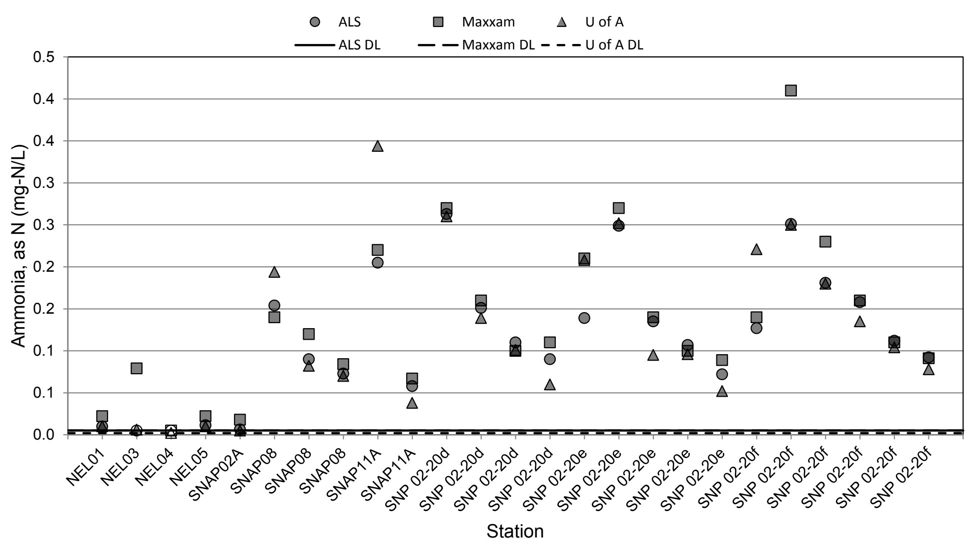
Maxxam, and were notably lower (i.e., by 0.5 mg/L or more) than both Maxxam and ALS in about a third of the split samples (Figure 12.4-10).

Similar to the results of the nitrate split samples, TKN results between ALS and Maxxam were generally more consistent with each other compared to TKN results from UofA. Approximately one third of the TKN values reported by the UofA were notably higher (i.e., by 0.5 mg/L or more) than those provided by ALS, and Maxxam (Figure 12.4-11). The source of the high bias in UofA values of TKN is related to the low bias in UofA's nitrate values because UofA calculates TKN by subtracting nitrate values from measured total nitrogen values (Equation 1 in Section 12.4.2.1 Spike Samples Preparation and Analysis).

The most consistent nitrogen data (for all forms of measured nitrogen) was provided by ALS, with the fewest noticeable differences between the ALS results, and the results from the other two laboratories. Additionally, the ALS results did not display a noticeable bias in any of the four forms of nitrogen.

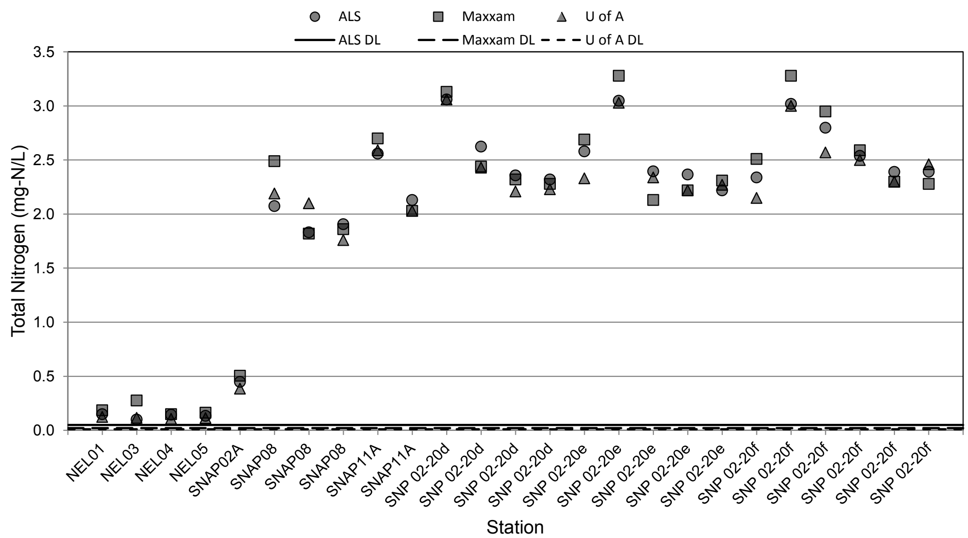
Results of TN were the least variable, relative to other TN values, among all three laboratories.

Figure 12.4-8 Split Samples Total Ammonia Laboratory Results



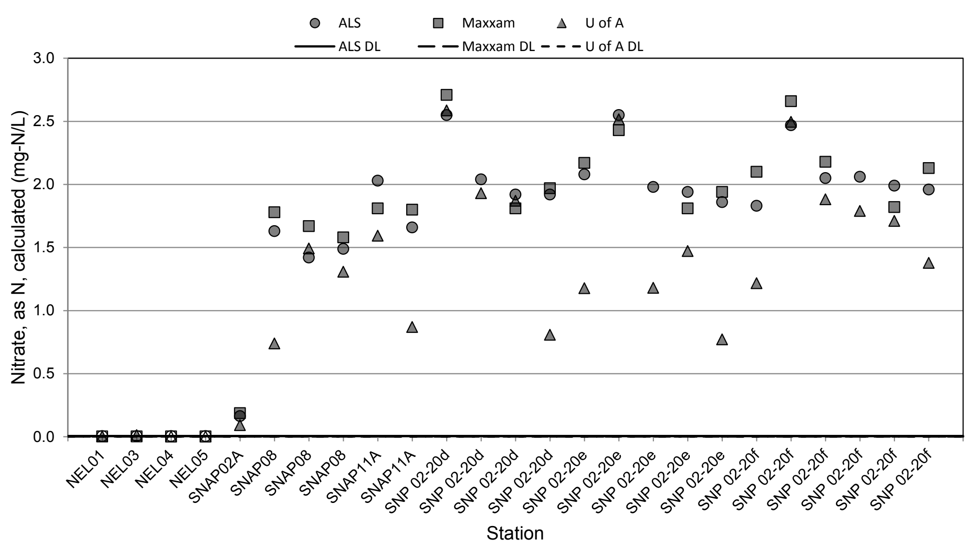
ALS = ALS Canada Ltd.; Maxxam = Maxxam Analytics Inc.; UofA = University of Alberta Biogeochemical Analytical Service Laboratory; DL = detection limit; mg-N/L = milligrams as nitrogen per litre; AEMP = Aquatic Effects Monitoring Program; NEL = Northeast Lake; SNP = Surveillance Network Program; ALS DL = 0.005 mg-N/L; Maxxam DL = 0.005 mg-N/L; UofA DL = 0.002 mg-N/L; open data points = laboratory results reported as less than detection limit; and samples collected from multiple dates are shown for AEMP stations SNAP08, SNAP11A, SNP 02-20d, SNP 02-20e, and SNP 02-20f.

Figure 12.4-9 Split Samples Total Nitrogen Laboratory Results



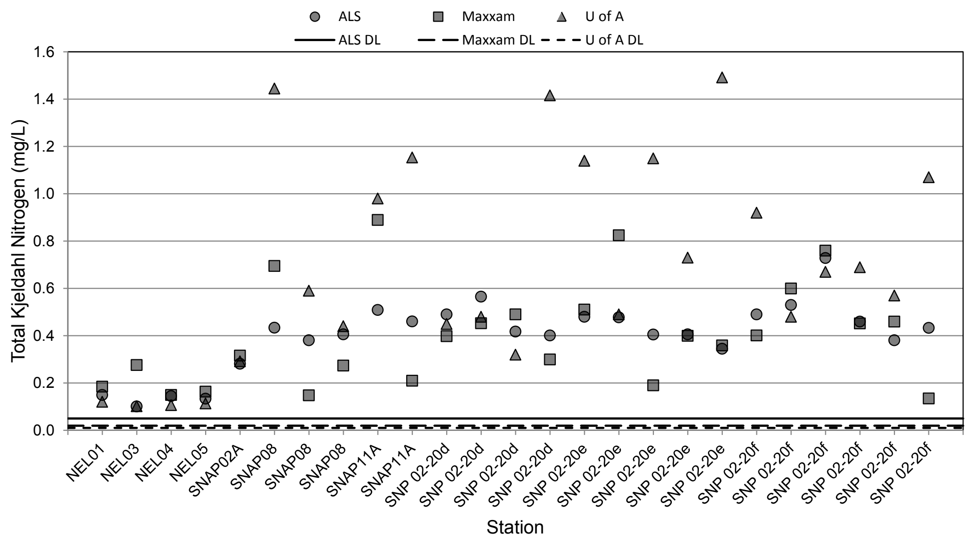
ALS = ALS Canada Ltd.; Maxxam = Maxxam Analytics Inc.; UofA = University of Alberta Biogeochemical Analytical Service Laboratory; DL = detection limit; mg-N/L = milligrams as nitrogen per litre; AEMP = Aquatic Effects Monitoring Program; NEL = Northeast Lake; SNP = Surveillance Network Program; ALS DL = 0.05 mg-N/L; Maxxam DL = 0.02 mg-N/L; UofA DL = 0.01 mg-N/L; open data points = laboratory results reported as less than detection limit; and samples collected from multiple dates are shown for AEMP stations SNAP08, SNAP11A, SNP 02-20d, SNP 02-20e, and SNP 02-20f.

Figure 12.4-10 Split Samples Nitrate Laboratory Results



ALS = ALS Canada Ltd.; Maxxam = Maxxam Analytics Inc.; UofA = University of Alberta Biogeochemical Analytical Service Laboratory; DL = detection limit; mg-N/L = milligrams as nitrogen per litre; AEMP = Aquatic Effects Monitoring Program; NEL = Northeast Lake; SNP = Surveillance Network Program; ALS DL = 0.006 mg-N/L; Maxxam DL = 0.002 mg-N/L; and UofA DL = 0.001 mg-N/L; open data points = laboratory results reported as less than detection limit; and samples collected from multiple dates are shown for AEMP stations SNAP08, SNAP11A, SNP 02-20d, SNP 02-20e, and SNP 02-20f.

Figure 12.4-11 Split Samples Total Kjeldahl Nitrogen Laboratory Results



ALS = ALS Canada Ltd.; Maxxam = Maxxam Analytics Inc.; UofA = University of Alberta Biogeochemical Analytical Service Laboratory; DL = detection limit; mg/L = milligrams per litre; AEMP = Aquatic Effects Monitoring Program; NEL = Northeast Lake; SNP = Surveillance Network Program; ALS DL = 0.05 mg/L; Maxxam DL = 0.02 mg/L; UofA DL = 0.01 mg/L; open data points = laboratory results reported as less than detection limit; and samples collected from multiple dates are shown for AEMP stations SNAP08, SNAP11A, SNP 02-20d, SNP 02-20e, and SNP 02-20f.

Phosphorus

The phosphorus concentrations of the split samples collected throughout the 2012 AEMP field program were low compared to the three laboratories' detection limits, with the majority of the phosphorus results less than five times the respective laboratory's detection limits. Due to uncertainties associated with low level phosphorus results (Section 12.4.2.1 Spike Samples Results and Discussion, Phosphorus), assessing discernible differences in the data sets provided by each laboratory was based on reported values greater than two times the detection limits of each laboratory. Two times the detection limit was selected because it is used for internal laboratory QA/QC acceptance criterion for Maxxam blank samples (Maxxam 2012). All results reported within two times the detection limit were considered to be equal.

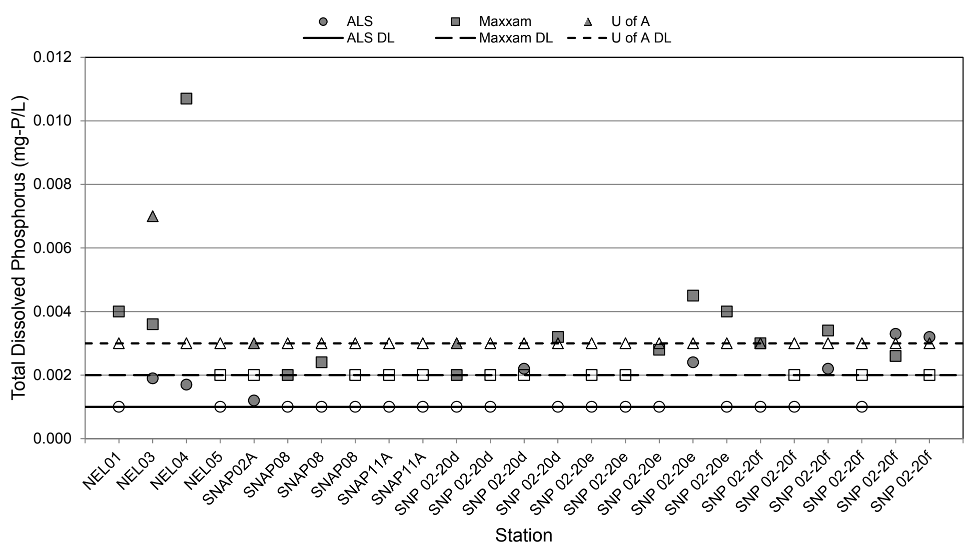
The TDP and ortho-phosphate results from split samples were generally similar between all three laboratories (Figures 12.4-12 and 12.4-13, respectively). The three laboratories reported TDP and ortho-phosphate at values less than, or equal to, their respective detection limits in just under half of the split samples. The UofA reported one TDP, and one ortho-phosphate value (from NEL03 July 10, 2012) that was more than two times the detection limit, and noticeably different (i.e., by more than 0.003 mg/L) than those reported by ALS, and Maxxam (Figures 12.4-12 and 12.4-13, respectively). Maxxam also reported one TDP value (from NEL04 July 10, 2012) that was more than two times the detection limit, and noticeably different than those reported by ALS and UofA (Figure 12.4-12).

The TP results from split samples were less consistent between the three laboratories compared to the TDP and the ortho-phosphate split results (Figure 12.4-14). Maxxam reported TP results greater than two times the detection limit that were notably greater (i.e., by more than 0.003 mg/L) than ALS and UofA TP results in two of the splits (NEL04 July 7, 2012 and SNP 02-20e February 19, 2012). Similarly, UofA also reported TP results greater than two times the detection limit that were notably greater (i.e., by more than 0.003 mg/L) than ALS, and UofA TP results in two of the splits (SNP 02-20e, and SNP 02-20f August 12, 2012). In addition, UofA reported a value of TP less than the detection limit in one of the splits (SNAP08 July 13, 2012), for which ALS and Maxxam reported TP values notably greater than the UofA detection limit.

Based on the split samples submitted for this study, ALS appeared to provide the most consistent low level phosphorus data relative to UofA and Maxxam because phosphorus results (i.e., TDP, ortho-phosphate, and TP) from ALS were not outliers in any of the split samples.

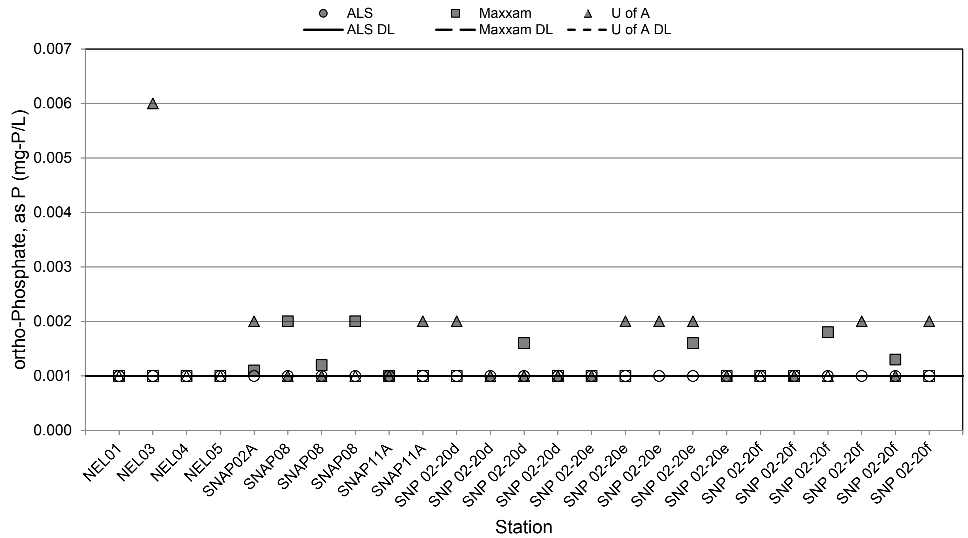
Overall, for both nitrogen and phosphorus, ALS provided the most consistent results in the split samples because they had the fewest number of results that were different than both UofA and Maxxam.

Figure 12.4-12 Split Samples Total Dissolved Phosphorus Laboratory Results



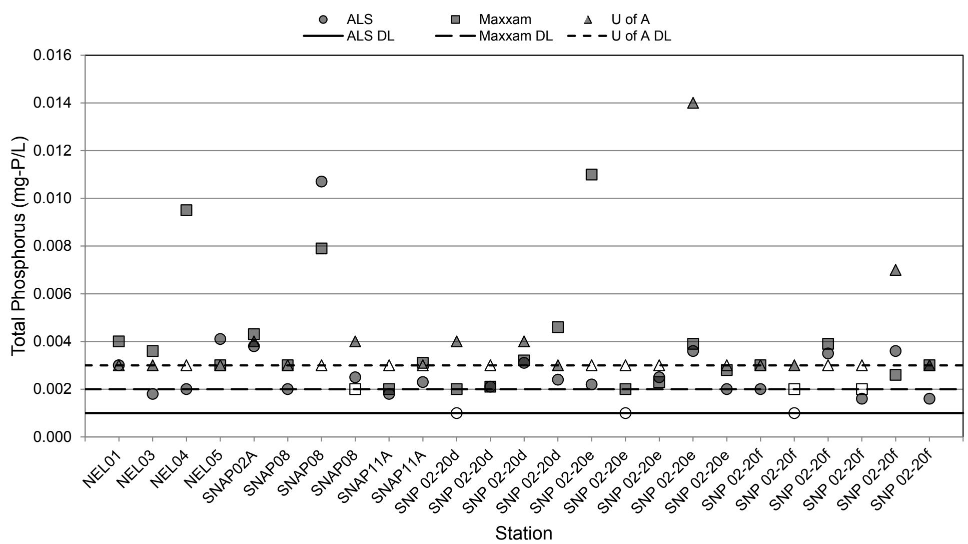
ALS = ALS Canada Ltd.; Maxxam = Maxxam Analytics Inc.; UofA = University of Alberta Biogeochemical Analytical Service Laboratory; DL = detection limit; mg-P/L = milligrams as phosphorus per litre; AEMP = Aquatic Effects Monitoring Program; NEL = Northeast Lake; SNP = Surveillance Network Program; ALS DL = 0.001 mg-P/L; Maxxam DL = 0.002 mg-P/L; UofA DL = 0.003 mg-P/L; open data points = laboratory results reported as less than detection limit; and Samples collected from multiple dates are shown for AEMP stations SNAP08, SNAP11A, SNP 02-20d, SNP 02-20e, and SNP 02-20f.

Figure 12.4-13 Split Samples ortho-Phosphate Laboratory Results



ALS = ALS Canada Ltd.; Maxxam = Maxxam Analytics Inc.; UofA = University of Alberta Biogeochemical Analytical Service Laboratory; DL = detection limit; mg-P/L = milligrams as phosphorus per litre; AEMP = Aquatic Effects Monitoring Program; NEL = Northeast Lake; SNP = Surveillance Network Program; ALS DL = 0.001 mg-P/L; Maxxam DL = 0.001 mg-P/L; UofA DL = 0.001 mg-P/L; open data points = laboratory results reported as less than detection limit; and Samples collected from multiple dates are shown for AEMP stations SNAP08, SNAP11A, SNP 02-20d, SNP 02-20e, and SNP 02-20f.

Figure 12.4-14 Split Samples Total Phosphorus Laboratory Results



ALS = ALS Canada Ltd.; Maxxam = Maxxam Analytics Inc.; UofA = University of Alberta Biogeochemical Analytical Service Laboratory; DL = detection limit; mg-P/L = milligrams as phosphorus per litre; AEMP = Aquatic Effects Monitoring Program; NEL = Northeast Lake; SNP = Surveillance Network Program; ALS DL = 0.001 mg-P/L; Maxxam DL = 0.002 mg-P/L; UofA DL = 0.003 mg-P/L; open data points = laboratory results reported as less than detection limit; and Samples collected from multiple dates are shown for AEMP stations SNAP08, SNAP11A, SNP 02-20d, SNP 02-20e, and SNP 02-20f.

12.4.3 Sample Collection Assessment

12.4.3.1 Key Question 3 - How do nutrient concentrations in mid-depth grab samples compare to depth-integrated euphotic zone composite samples collected at the same station?

Sample Collection and Analysis

To determine whether sampling depth influences the concentration of nutrients, the results of the samples collected at mid-depth, and the depth-integrated euphotic zone composite samples (euphotic zone samples) collected at the same stations were compared for differences in nutrient concentration. This key question focused only on the sample collection depth, comparing the nutrient concentrations of paired samples provided by each laboratory. It was assumed that the specific biases identified in the previous sections would be consistent within each laboratory, and thereby affect samples from different depths equally.

During the September 2012 field program, samples were collected at mid-depth and within the euphotic zone from seven stations (Table 12.4-11). The samples were collected in triplicate, and sent to the three laboratories for the nutrient analyses described in Section 12.4.2.1 (Spike Samples Preparation and Analysis).

Table 12.4-11 2012 Nutrient Study Sample Depth Comparison Stations

Station	Location Description	Collection Date
NEL02	Northeast Lake	2012-09-08
NEL04		2012-09-08
SNAP02A	northwest arm in Snap Lake	2012-09-12
SNAP08	far-field area near the outlet of Snap Lake	2012-09-11
SNP 02-20d	diffuser stations in the near-field area of Snap Lake	2012-09-14
SNP 02-20e		2012-09-14
SNP 02-20f		2012-09-14

Note: date format is YYYY-MM-DD, where Y= year, M = month and D = day.

The water for the mid-depth samples was collected following the open-water season procedures described in Section 12.4.2.1 (Spike Samples Preparation and Analysis).

For the euphotic zone samples, water was collected from the top 6 metres (m) of the water column at each of the stations. The depth of the euphotic zone was determined following the procedures described in Section 5.2.1.5 (Sampling Methods). Water was collected using a 2 L Kemmerer water sampler at 2 m intervals to a maximum of 6 m (i.e., just below the water surface [0.1 to 0.2 m], 2 m, 4 m, and 6 m). Equal volumes of water collected at each depth were mixed in a clean 11 L bucket to generate a composite sample, which was then transferred into three sets of appropriate sampling bottles, one set for each laboratory.

All mid-depth and euphotic zone samples were to be analyzed for the parameters listed in Section 12.4.2.1 (Spike Samples Preparation and Analysis); however, due to issues related to shipping, missing bottles, and sample preservation, not all samples were analyzed for all parameters.

Data Analyses

The data analyses focused on ammonia, nitrate, ortho-phosphate, TDP, TKN, TN, and TP. Results from the seven stations were assessed for differences in nutrient concentration. The relative percent difference (RPD) of the paired results from each laboratory was calculated to determine whether there were differences in the nutrient concentration between the two sampling depths. An acceptance criterion of less than 20% was used throughout the assessment. Relative percent difference was calculated using the following equation:

$$\text{Relative Percent Difference} = \frac{(\text{middepth concentration} - \text{euphotic zone concentration})}{(\text{middepth concentration} + \text{euphotic zone concentrationd}) \div 2} \times 100 \quad \text{[Equation 12-5]}$$

Results and Discussion

Laboratory results from the mid-depth, and euphotic zone samples from the same station were used to determine whether the sampling depths of the water quality and plankton component of the AEMP could be streamlined to one consistent sampling depth. Streamlining the water quality, and plankton sampling procedures was a recommendation in the 2013 AEMP Design Update (De Beers 2012), pending the results of this study.

Nitrogen

Concentrations of ammonia were generally consistent between the two sampling depths (Figure 12.4-15 and Table 12.4-12). When samples were identified as notably different, that is when the RPD between the mid-depth, and euphotic depth samples was greater than 20%, the euphotic zone samples had the highest concentrations of ammonia.

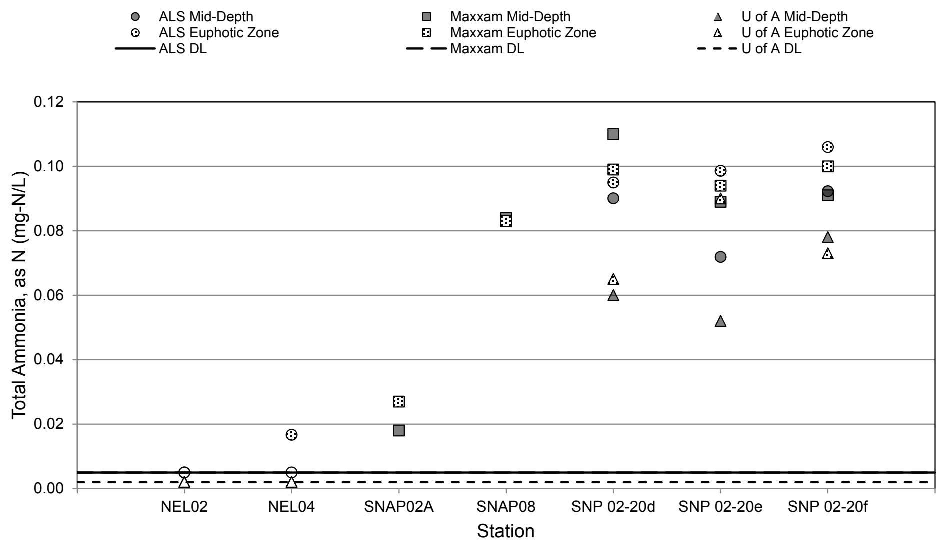
Nitrate concentrations were also generally consistent between the two sampling depths (Figure 12.4-16 and Table 12.4-12). Approximately one third of the samples had higher nitrate concentrations in the euphotic zone samples, and one third had higher concentrations in the mid-depth samples.

Concentrations of TKN were generally higher in the euphotic zone samples than the mid-depth samples (Figure 12.4-17 and Table 12.4-12). In all instances where the RPD between the two samples was greater than 20%, the euphotic zone samples had higher concentrations of TKN, with the exception of one instance where the UofA reported a higher value in the mid-depth sample.

Concentrations of TN were consistent between the two sampling depths within Snap Lake. The samples collected from Northeast Lake had noticeably higher TN concentrations in the euphotic zone samples than in the mid-depth samples (Figure 12.4-18 and Table 12.4-12). All four paired results collected from Northeast Lake had a RPD greater than 20%.

The concentrations of TN were the most consistent, relative to other nitrogen parameters, between the two sampling depths. The average RPD in samples for TN was 18%, compared to 20% for ammonia, 22% for nitrate, and 39% for TKN (Table 12.4-12). All paired nitrogen results collected from Northeast Lake had a RPD greater than 20%, with the exception of the three pairs of results that were all below detection limits, and therefore the actual RPD could not be calculated (Table 12.4-12).

Figure 12.4-15 Mid-depth and Euphotic Zone Samples Total Ammonia Laboratory Results



ALS = ALS Canada Ltd.; Maxxam = Maxxam Analytics Inc.; UofA = University of Alberta Biogeochemical Analytical Service Laboratory; DL = detection limit; mg-N/L = milligrams as nitrogen per litre; NEL = Northeast Lake; SNP = Surveillance Network Program; ALS DL = 0.005 mg-N/L; Maxxam DL = 0.005 mg-N/L; and UofA DL = 0.002 mg-N/L; and open data points = laboratory results reported as less than detection limit.

Table 12.4-12 Nitrogen Results of Mid-depth and Euphotic Zone Samples

Parameter and Station	Laboratory	Mid-Depth Concentration (mg-N/L)	Euphotic Zone Concentration (mg-N/L)	Relative Percent Difference (%)[a]
Nitrate, as N, calculated[b]				
NEL02	UofA	0.002	0.003	**40**
NEL04	UofA	0.003	0.001	**100**
SNAP02A	Maxxam	0.187	0.211	12
SNAP08	Maxxam	1.580	1.580	0
SNP 02-20d	ALS	1.920	1.840	4
	Maxxam	1.970	1.970	0
	UofA	0.808	0.943	15
SNP 02-20e	ALS	1.860	1.890	2
	Maxxam	1.940	1.940	0

Table 12.4-12 Nitrogen Results of Mid-depth and Euphotic Zone Samples

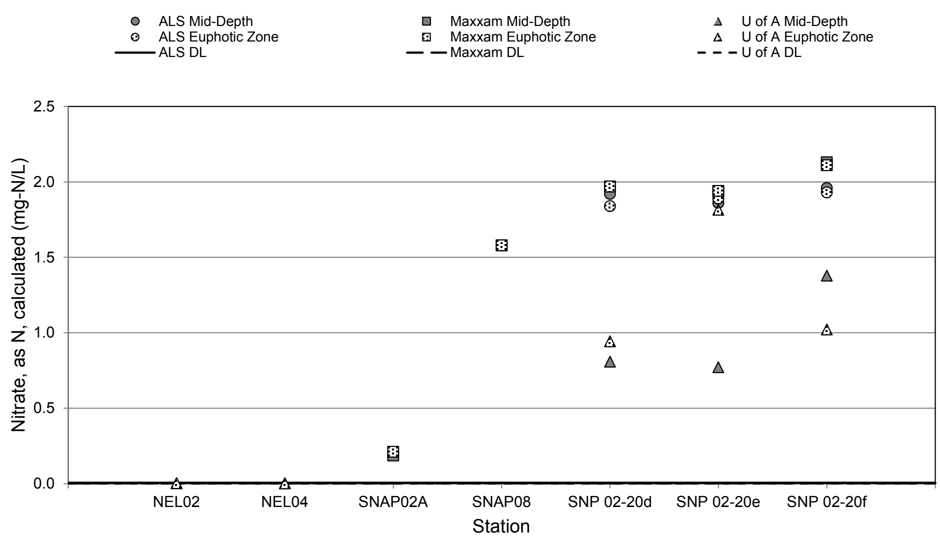
Parameter and Station	Laboratory	Mid-Depth Concentration (mg-N/L)	Euphotic Zone Concentration (mg-N/L)	Relative Percent Difference (%)[(a)]
	UofA	0.771	1.815	**81**
SNP 02-20f	ALS	1.960	1.930	2
	Maxxam	2.130	2.110	1
	UofA	1.378	1.021	**30**
			Average:	**22**
Total Ammonia, as N				
NEL02	ALS	<0.005	<0.005	0
	UofA	<0.002	<0.002	0
NEL04	ALS	<0.005	0.017	**108**
	UofA	<0.002	<0.002	0
SNAP02A	Maxxam	0.018	0.027	**40**
SNAP08	Maxxam	0.084	0.083	1
SNP 02-20d	ALS	0.090	0.095	5
	Maxxam	0.110	0.099	11
	UofA	0.060	0.065	8
SNP 02-20e	ALS	0.072	0.099	**31**
	Maxxam	0.089	0.094	5
	UofA	0.052	0.090	**54**
SNP 02-20f	ALS	0.092	0.106	14
	Maxxam	0.091	0.100	9
	UofA	0.078	0.073	7
			Average:	20
Total Kjeldahl Nitrogen				
NEL02	ALS	0.13	0.19	**41**
	UofA	0.14	0.22	**43**
NEL04	ALS	0.10	0.28	**95**
	UofA	0.12	0.21	**50**
SNP 02-20d	ALS	0.40	0.45	12
	Maxxam	0.30	0.36	17
	UofA	1.42	1.30	9
SNP 02-20e	ALS	0.35	0.46	**29**
	Maxxam	0.36	0.38	7
	UofA	1.49	0.47	**104**
SNP 02-20f	ALS	0.43	0.47	8
	Maxxam	0.14	0.28	**71**
	UofA	1.07	1.38	**25**
			Average:	**39**
Total Nitrogen[(c)]				
NEL02	ALS	0.13	0.19	**41**
	UofA	0.14	0.22	**43**
NEL04	ALS	0.10	0.28	**95**
	UofA	0.13	0.21	**49**
SNAP02A	UofA	0.39	0.46	18
SNAP08	UofA	1.76	1.80	2
SNP 02-20d	ALS	2.32	2.31	0
	Maxxam	2.28	2.34	3
	UofA	2.23	2.25	1

Table 12.4-12 Nitrogen Results of Mid-depth and Euphotic Zone Samples

Parameter and Station	Laboratory	Mid-Depth Concentration (mg-N/L)	Euphotic Zone Concentration (mg-N/L)	Relative Percent Difference (%)[(a)]
SNP 02-20e	ALS	2.22	2.35	6
	Maxxam	2.31	2.34	1
	UofA	2.27	2.30	1
SNP 02-20f	ALS	2.39	2.40	0
	Maxxam	2.28	2.41	6
	UofA	2.46	2.41	2
			Average:	18

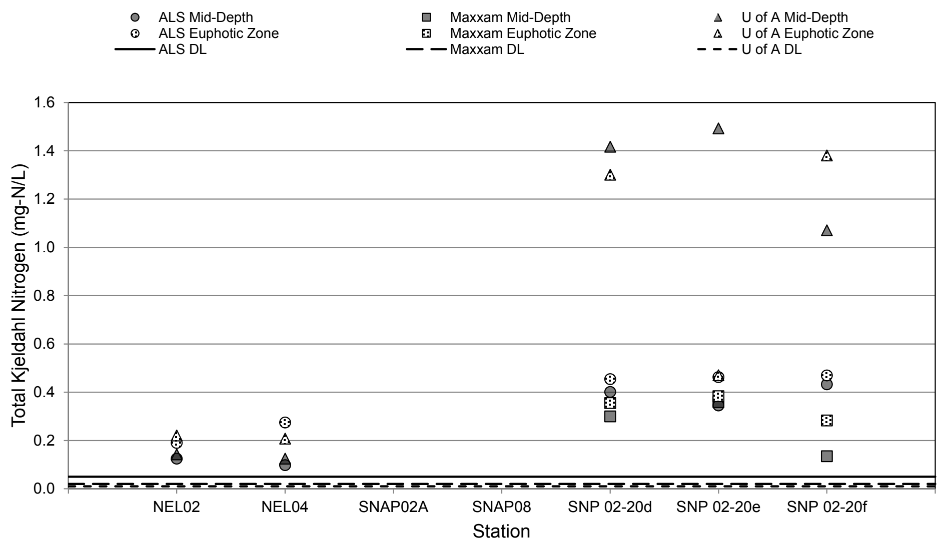
Note: **Bold** indicates RPD value outside acceptance criteria of 20%.
(a) If the laboratory value was reported as less than the detection limit, the respective laboratory's detection limit was used for calculation purposes.
(b) Calculated UofA nitrate concentrations were equal to the combined measured nitrate and nitrite concentrations minus the measured nitrite concentrations.
(c) ALS TN values were calculated as the sum of measured TKN and measured combined nitrate and nitrite values.
Abbreviations: ALS = ALS Canada Ltd.; Maxxam = Maxxam Analytics Inc.; UofA = University of Alberta Biogeochemical Analytical Service Laboratory; mg = milligram; N= nitrogen; L = litre; RPD = relative percent difference; % = percent; < = less than; TKN = total Kjeldahl nitrogen; TN = total nitrogen.

Figure 12.4-16 Mid-depth and Euphotic Zone Nitrate Laboratory Results



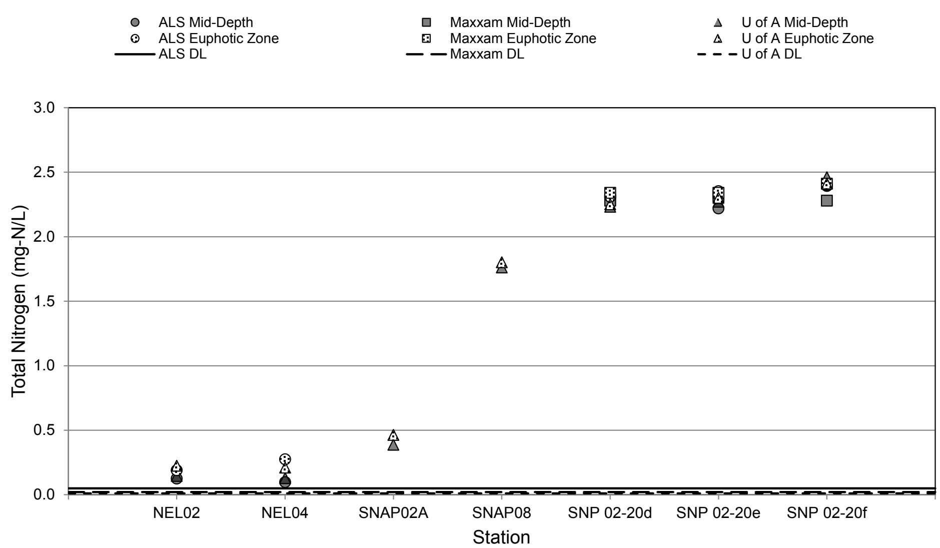
ALS = ALS Canada Ltd.; Maxxam = Maxxam Analytics; UofA = University of Alberta Biogeochemical Analytical Service Laboratory; DL = detection limit; mg-N/L = milligrams as nitrogen per litre; NEL = Northeast Lake; SNP = Surveillance Network Program.; ALS DL = 0.006 mg-N/L; Maxxam DL = 0.002 mg-N/L; and UofA DL = 0.001 mg-N/L; and open data points = laboratory results reported as less than detection limit.

Figure 12.4-17 Mid-depth and Euphotic Zone Samples Total Kjeldahl Nitrogen Laboratory Results



ALS = ALS Canada Ltd.; Maxxam = Maxxam Analytics Inc.; UofA = University of Alberta Biogeochemical Analytical Service Laboratory; DL = detection limit; mg-N/L = milligrams as nitrogen per litre; NEL = Northeast Lake; SNP = Surveillance Network Program; ALS DL = 0.05 mg/L; Maxxam DL = 0.02 mg/L; UofA DL = 0.01 mg/L; and open data points = laboratory results reported as less than detection limit.

Figure 12.4-18 Mid-depth and Euphotic Zone Samples Total Nitrogen Laboratory Results



ALS = ALS Canada Ltd.; Maxxam = Maxxam Analytics Inc.; UofA = University of Alberta Biogeochemical Analytical Service Laboratory; DL = detection limit; mg-N/L = milligrams as nitrogen per litre; NEL = Northeast Lake; SNP = Surveillance Network Program; ALS DL = 0.05 mg-N/L; Maxxam DL = 0.02 mg-N/L; UofA DL = 0.01 mg-N/L; and open data points = laboratory results reported as less than detection limit.

Phosphorus

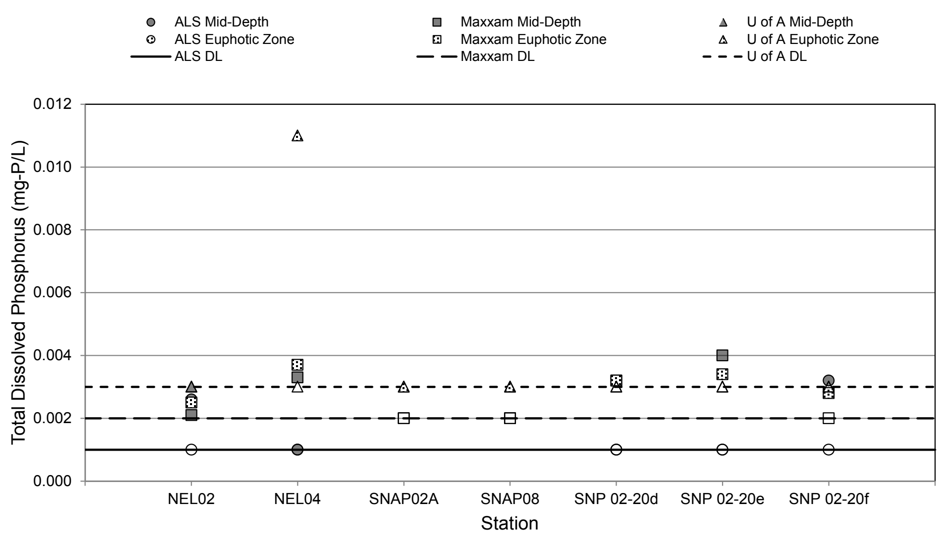
Concentrations of TDP were generally consistent between the two sampling depths (Figure 12.4-19 and Table 12.4-13). Half of the results with RPDs greater than 20% had higher TDP concentrations in the euphotic zone samples, and half had higher concentrations in the mid-depth samples. Approximately two thirds of the paired results had a RPD equal to zero, and all reported TDP concentrations from both the euphotic zone samples, and mid-depth samples were less than five times the respective laboratory detection limit.

Ortho-phosphate concentrations were also generally consistent between the two sampling depths; all but two paired results (Figure 12.4-20 and Table 12.4-13) had an RPD equal to zero. The two paired results that did have a RPD greater than 20% had higher concentrations in the euphotic zone samples. Due to field sampling issues ortho-phosphate was not analysed for all samples (i.e., not for NEL02, NEL04, SNAP02A, and SNAP08) or by all laboratories.

Concentrations of TP were higher in the euphotic zone samples than the mid-depth samples (Figure 12.4-21 and Table 12.4-13). In all instances where the RPD between the two samples was greater than 20%, the euphotic zone samples had higher concentrations of TP.

Overall, TDP and ortho-phosphate concentrations were similar at both sampling depths, and TP concentrations were higher in the euphotic zone samples compared to the corresponding mid-depth samples.

Figure 12.4-19 Mid-depth and Euphotic Zone Samples Total Dissolved Phosphorus Laboratory Results



ALS = ALS Canada Ltd.; Maxxam = Maxxam Analytics Inc.; UofA = University of Alberta Biogeochemical Analytical Service Laboratory; DL = detection limit; mg-P/L = milligrams as phosphorus per litre; NEL = Northeast Lake; SNP = Surveillance Network Program; ALS DL = 0.001 mg-P/L; Maxxam DL = 0.002 mg-P/L; UofA DL = 0.003 mg-P/L; and open data points = laboratory results reported as less than detection limit.

Table 12.4-13 Phosphorus Results of Mid-depth and Euphotic Zone Samples

Parameter and Station	Laboratory	Mid-Depth Concentration (mg-P/L)	Euphotic Zone Concentration (mg-P/L)	Relative Percent Difference (%)[a]
ortho-Phosphate, as P				
NEL02	UofA	<0.001	<0.001	0
NEL04	UofA	<0.001	<0.001	0
SNP 02-20d	ALS	<0.0010	<0.0010	0
	Maxxam	<0.001	<0.001	0
	UofA	0.001	0.002	**67**
SNP 02-20e	ALS	<0.0010	<0.0010	0
	Maxxam	<0.001	<0.001	0
	UofA	0.001	0.002	**67**
SNP 02-20f	ALS	<0.0010	<0.0010	0
	Maxxam	<0.001	<0.001	0
	UofA	0.002	0.002	0
			Average:	12
Total Dissolved Phosphorus				
NEL02	ALS	0.0026	<0.001	**89**
	Maxxam	0.0021	0.0025	17
	UofA	0.003	<0.003	0
NEL04	ALS	0.001	<0.001	0
	Maxxam	0.0033	0.0037	11
	UofA	<0.003	0.011	**114**
SNAP02A	Maxxam	<0.002	<0.002	0
	UofA	0.003	0.003	0
SNAP08	Maxxam	<0.002	<0.002	0
	UofA	<0.003	0.003	0
SNP 02-20d	ALS	<0.0010	<0.0010	0
	Maxxam	0.0032	0.0032	0
	UofA	<0.003	<0.003	0
SNP 02-20e	ALS	<0.0010	<0.0010	0
	Maxxam	0.004	0.0034	16
	UofA	<0.003	<0.003	0
SNP 02-20f	ALS	0.0032	<0.001	**105**
	Maxxam	<0.002	0.0028	**33**
	UofA	<0.003	<0.003	0
			Average:	20
Total Phosphorus				
NEL02	ALS	0.0032	0.009	**95**
	UofA	0.005	0.008	**46**
NEL04	ALS	0.003	0.0185	**144**
	UofA	0.003	0.017	**140**
SNAP02A	Maxxam	0.0043	0.0072	**50**
	UofA	0.004	0.015	**116**
SNAP08	Maxxam	<0.002	0.0025	**22**
	UofA	0.004	0.006	**40**
SNP 02-20d	ALS	0.0024	0.0031	**25**
	Maxxam	0.0046	0.005	8
	UofA	0.003	0.004	**29**

Table 12.4-13 Phosphorus Results of Mid-depth and Euphotic Zone Samples

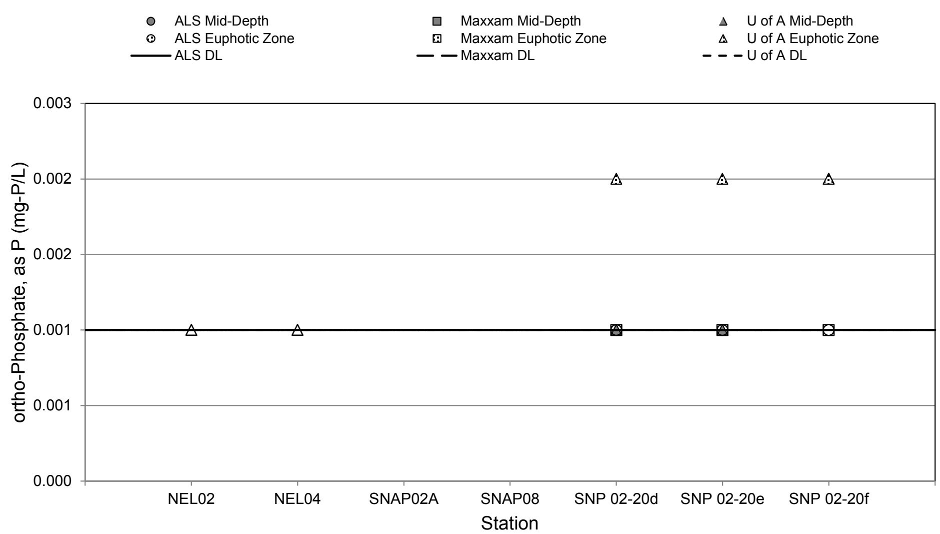
Parameter and Station	Laboratory	Mid-Depth Concentration (mg-P/L)	Euphotic Zone Concentration (mg-P/L)	Relative Percent Difference (%)(a)
SNP 02-20e	ALS	0.002	0.0028	**33**
	Maxxam	0.0028	0.007	**86**
	UofA	<0.003	0.004	**29**
SNP 02-20f	ALS	0.0016	0.0022	**32**
	Maxxam	0.003	0.0025	18
	UofA	0.003	0.004	**29**
			Average:	55

Note: **Bold** indicates RPD value outside acceptance criteria of 20%.

(a) If the laboratory value was reported as less than the detection limit, the respective laboratory's detection limit was used for calculation purposes.

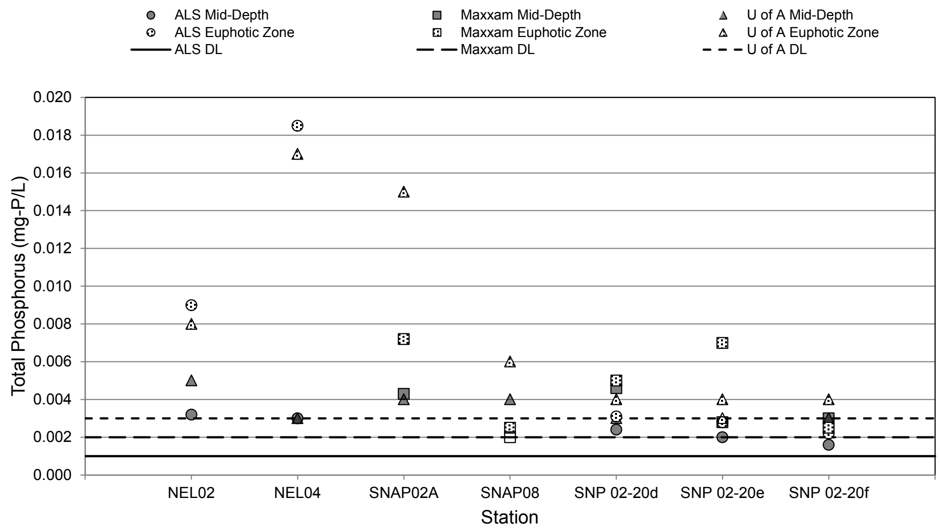
ALS = ALS Canada Ltd.; Maxxam = Maxxam Analytics Inc.; UofA = University of Alberta Biogeochemical Analytical Service Laboratory; mg-P/L = milligrams as phosphorus per litre; RPD = relative percent difference; % = percent; < = less than.

Figure 12.4-20 Mid-depth and Euphotic Zone Samples ortho-Phosphate Laboratory Results



ALS = ALS Canada Ltd.; Maxxam = Maxxam Analytics Inc.; UofA = University of Alberta Biogeochemical Analytical Service Laboratory; DL = detection limit; mg-P/L = milligrams as phosphorus per litre; AEMP = aquatics effects monitoring program; NEL = Northeast Lake; SNP = Surveillance Network Program; ALS DL = 0.001 mg-P/L; Maxxam DL = 0.001 mg-P/L; UofA DL = 0.001 mg-P/L; open data points = laboratory results reported as less than detection limit; and ortho-phosphate not analyzed for AEMP stations SNAP02A and SNAP08.

Figure 12.4-21 Mid-depth and Euphotic Zone Samples Total Phosphorus Laboratory Results



ALS = ALS Canada Ltd.; Maxxam = Maxxam Analytics Inc.; UofA = University of Alberta Biogeochemical Analytical Service Laboratory; DL = detection limit; mg-P/L = milligrams as phosphorus per litre; NEL = Northeast Lake; SNP = Surveillance Network Program; ALS DL = 0.001 mg-P/L; Maxxam DL = 0.002 mg-P/L; UofA DL = 0.003 mg-P/L; and open data points = laboratory results reported as less than detection limit.

12.4.4 Study Limitations

The 2012 Nutrient Special Study was designed to further investigate possible reasons for the inconsistencies observed between the nutrient data collected through the water quality, and plankton programs in the Snap Lake AEMP. Based on the limited number of spike, and split samples collected, and analyzed for the study, statistical analysis of the results was not undertaken. However, the available data were considered sufficient to evaluate the reliability of the three laboratories used for the AEMP, and to identify obvious patterns in data that may indicate laboratory issues or trends in water quality, which could then be investigated further if required.

Due to the required clean nature of the matrix of the spike samples, the spike samples used to assess the accuracy of laboratory results were not meant to represent the complete sample matrix present in a water sample collected from Snap Lake. Therefore, the level of accuracy observed in the spike samples may not be equivalent to the level of accuracy in AEMP results due to the possibility of potential analytical interferences in the lake waters. However, the results for the spike samples can be used as an indication of a laboratory's ability to accurately measure nutrient concentrations for the AEMP.

12.4.5 Key Findings

12.4.5.1 Key Question 1: Are the Laboratories Able to Accurately Measure Known Concentrations of Nutrients?

The results of the spike samples generated from certified standard material did not differentiate UofA, Maxxam, or ALS in terms of their ability to measure nutrients accurately. In general, the three laboratories provided accurate nitrogen results for the range of concentrations of the spike samples, with the possible exception of the TKN results provided by Maxxam that suggest a high bias. Maxxam provided the most accurate phosphorus results; however, the results of the phosphorus analyses from ALS and UofA may be skewed because both laboratories grossly overestimated the phosphorus concentration in one spike sample.

The three laboratories were the most accurate at measuring higher phosphorus concentrations (0.01 mg-P/L) compared to lower phosphorus concentrations (0.0015 mg-P/L), which is below the detection limits for Maxxam and the UofA.

12.4.5.2 Key Question 2: Are There Patterns in Differences in the Nutrient Data Provided by Each Laboratory?

Overall, ALS provided the most consistent nitrogen and phosphorus data, with the fewest noticeable differences from the other two laboratories. The UofA results for nitrate were typically lower than those provided by ALS and Maxxam, which indicates that UofA may have underestimated nitrate results in the AEMP split samples. Because UofA subtracts nitrate from TN to calculate TKN, TKN values from UofA splits samples appeared to be biased high. Phosphorus concentrations in split samples were typically low and similar to each other for all laboratories, particularly for TDP and ortho-phosphate, which were often below detection limits. Because concentrations of total phosphorus were typically above detection limits, more variability between the three laboratories was observed. In split samples where concentrations differed, the laboratories with the most noticeable differences were UofA and Maxxam.

12.4.5.3 Key Question 3: How Do Nutrient Concentrations in Mid-Depth Grab Samples Compare to Euphotic Zone Composite Samples Collected at the Same Station?

The results of the comparisons of nutrient samples collected at different depths in Snap Lake, and Northeast Lake indicated that nutrient concentrations may be different at different depths. Where noticeable differences were measured for ammonia, nitrate, and TKN, concentrations in the euphotic zone were generally higher than the mid-depth concentration. Concentrations of TN were consistent between the two sampling depths in Snap Lake; however, total nitrogen concentrations in Northeast Lake were higher in the euphotic zone samples compared to the corresponding mid-depth samples.

Total phosphorus concentrations were higher in the euphotic zone samples compared to the corresponding mid-depth samples. Ortho-phosphate and TDP concentrations were similar at both sampling depths.

12.4.6 Recommendations

The results of the laboratory assessment indicate that there is no clear choice for a preferred laboratory because the percent error results from UofA, Maxxam, and ALS in the spike samples did not differentiate any one laboratory in terms of overall accuracy. Therefore, it is recommended to continue to use ALS as the primary laboratory for the water quality component of the AEMP. The results of the split samples reinforce this recommendation, because ALS provided the most consistent nutrient data, with the fewest noticeable differences to results from the other two laboratories. Continued split sampling, which is part of the regular QA/QC procedures in the AEMP, is recommended to provide an external check of the primary laboratories completing the analyses. A limited number of nutrient spike samples should routinely be sent to UofA, Maxxam, and ALS as an on-going and independent check of the accuracy of nutrient results from all three laboratories.

The lack of accuracy in low-level phosphorus results in the spike samples should be considered when interpreting trends in phosphorus data in Snap Lake, thus establishing management action levels for phosphorus, and developing nutrient models for Snap Lake. Existing efforts to reduce uncertainty in low-level phosphorus, such as reducing field contamination of samples through documented QA/QC procedures, should continue.

Because the results of nutrient samples collected at different depths demonstrate that nutrient concentrations, particularly total phosphorus, may vary with sampling depth, reduction to a single combined sampling depth for water quality, and plankton components is not recommended at this time. Additional nutrient samples should be collected at mid-depth and in the euphotic zone to better define which forms of nutrients differ with sample depth, and the degree to which this difference may affect other nutrient-related components, and activities at Snap Lake, such as benthic invertebrates, and water quality modelling.

12.4.7 References

ALS (ALS Canada Ltd.). 2012. ALS Quality Control Protocols. v03. Winnipeg, MB, Canada.

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13 QUALITATIVE INTEGRATION

The Environmental Assessment Report (EAR; De Beers 2002) for the Snap Lake Mine (Mine) predicted inputs of nutrients, metals, and major ions to Snap Lake that could result in a combination of enrichment, resulting in mild stimulation (considered likely), and toxicity, resulting in impairment (considered unlikely), of the biological communities in Snap Lake. The component sections of the annual Aquatic Effects Monitoring Program (AEMP) report are designed to individually characterize changes in measures of contaminant and nutrient exposure, potential receiving water toxicity, and any resulting biological responses by plankton, benthos, and fish. Changes in these individual components could have a combined or interactive effect on the aquatic ecosystem of Snap Lake, which is the focus of the Qualitative Integration component.

Schedule 6, Part G, Conditions Applying to Aquatic Effects Monitoring of the Water License MV2011L2-0004 (MVLWB 2012), Section 4f states that the AEMP Annual report shall include:

> *"an analysis that integrates the results of individual monitoring components collected in a calendar year and describes the ecological significance of the results".*

The purpose of this section is to satisfy this requirement by conducting a qualitative integration of the measures of contaminant and nutrient exposure, and biological response described in the findings of the AEMP Component Sections. The qualitative integration follows weight of evidence (WOE) principles as described in the scientific literature (e.g., as described by Chapman and Anderson 2005; McDonald et al. 2007) and provincial, and federal guidance in Canada (e.g., SAB 2008; Environment Canada and Ontario Ministry of the Environment 2008; Azimuth 2012). The qualitative integration examines the relative likelihoods that enrichment effects and toxicity effects are happening in Snap Lake.

13.1 APPROACH

The WOE is defined as "any process used to aggregate information from different lines of scientific evidence to render a conclusion regarding the probability and magnitude of harm" (Azimuth 2012). This definition encompasses a range of practice, ranging from best professional judgment assessments to complex quantitative methods (Azimuth 2012). It is a well-established and accepted method for integrating environmental assessment data (e.g., Chapman and Anderson 2005; McDonald et al. 2007; Chapman and Smith 2012), and guidance on WOE methods have been developed and are in use in Canada both provincially (e.g., SAB 2008; Environment Canada and Ontario Ministry of the Environment 2008) and federally (Azimuth 2012).

A WOE Approach has been proposed as part the Draft AEMP Design Plan for Snap Lake (De Beers 2012), under the Water License Renewal process. The proposed WOE Approach includes a transition from a qualitative to a semi-quantitative approach as the findings of successive monitoring years are obtained under the new design. Because the current AEMP for 2012 was not designed to specifically support a WOE assessment, and is currently in transition, the current assessment has been limited to a qualitative integration of the various endpoints, conducted in alignment with the proposed WOE Approach wherever possible.

In general terms, in the qualitative integration, the endpoint results for each AEMP component are *rated* according to a series of decision criteria, *weighted qualitatively* to reflect the strength and relevance of the evidence they bring to the assessment, and then *integrated* to provide an overall qualitative integration indicating the degree of support for alternative hypotheses regarding the type of effect in Snap Lake. The approach includes the following features and considerations:

- It is designed to indicate the relative degree of support that the AEMP findings provide for two alternative hypotheses: *nutrient enrichment* versus *toxicological impairment*.
- Each hypothesis is examined for each broad ecosystem component within Snap Lake: plankton community, benthic invertebrate community, and fish community.
- Exposure and biological response endpoints are considered together with the overall findings for each type of endpoint to provide an integration of exposure and biological response.
- The quantitative and qualitative findings for each AEMP component are rated according to a standard set of rating guidelines which considers the magnitude, direction, and extent of responses in these endpoints. Application of these ratings errs on the side of caution (i.e., in the direction of a false-positive) to represent the potential worst-case responses in the component endpoints.
- The *representativeness* of each endpoint (i.e., how well it can indicate potential effects or changes in Snap Lake) and endpoint group is considered through a qualitative weighting which is based on published literature, guidance, and best professional judgement.
- It integrates this information in a qualitative fashion; i.e., a side-by-side presentation of exposure, and biological response endpoints to determine the degree of support for each hypothesis.

Additional detail regarding these steps and considerations are provided in Sections 13.1.1 to 13.1.5.

13.1.1 Conceptual Model and Hypotheses

Conceptual site models illustrate potential interactions of stressors of potential concern, exposure pathways, and receptors of potential concern. A detailed conceptual model is provided in the Draft AEMP Design Plan (De Beers 2012), and a brief overview focussed on components relevant to the qualitative integration is provided below.

The Mine-related stressors of potential concern relevant to qualitative integration for Snap Lake are:

- total dissolved solids (TDS) and its constituent ions;
- metals[11];
- the nutrients phosphorus (P) and nitrogen (N); and,
- acidifying substances.

The major source of TDS, associated ions, and metals to Snap Lake is groundwater that enters the mine workings, which is collected and directed to the water treatment plant, and is discharged to Snap Lake following treatment. Additional potential minor sources of these substances are seepages, spills, uncontrolled runoff, and dust deposition. The sources of nutrients in Snap Lake are: (i) nitrogen in explosive residues which enter groundwater seeping into the Mine, runoff waters, or treated domestic waste water, and possibly seep directly into the lake; and, (ii) phosphorus mainly in treated domestic waste water, and potentially in surface runoff.

The EAR also determined that acid deposition is a concern primarily for small inland lakes and small streams, and less so for Snap Lake because the discharge to Snap Lake contributes additional alkalinity, making it less acid-sensitive over time. As a result, acid deposition was not included in the qualitative integration but is addressed in Section 3 (Water Quality).

Based on the review of sources and pathways in the EAR (De Beers 2002), and on the clear relationships shown in AEMP data between concentrations of chemicals of potential concern in lake water and their concentrations, and loading rates in treated effluent. The primary exposure route for receptors of potential concern in Snap Lake is via the treated effluent discharge.

Receptors of potential concern are the following broad components of the Snap Lake ecosystem:

- primary producers (periphyton and phytoplankton communities);
- zooplankton;
- benthic invertebrates;
- demersal and pelagic fish; and,
- humans (indirectly through resource use).

Of these, phytoplankton, zooplankton, benthic invertebrates, and fish are included in the qualitative integration because represent direct effects to the biological community of Snap Lake. Periphyton are currently the subject of special studies but are not yet included in standard AEMP monitoring.

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

The pathways by which the above-identified sources may influence the aquatic ecosystem are both direct and indirect. Direct pathways involve a direct influence on a receptor, for example, direct toxicity to fish as a result of the elevated concentration of an ion or a metal. Indirect pathways often include several levels of receptors; for example, sediment input causing a reduction in benthic invertebrate density, thereby reducing the amount of food available for fish, is a scenario that includes both benthic invertebrate and fish receptors.

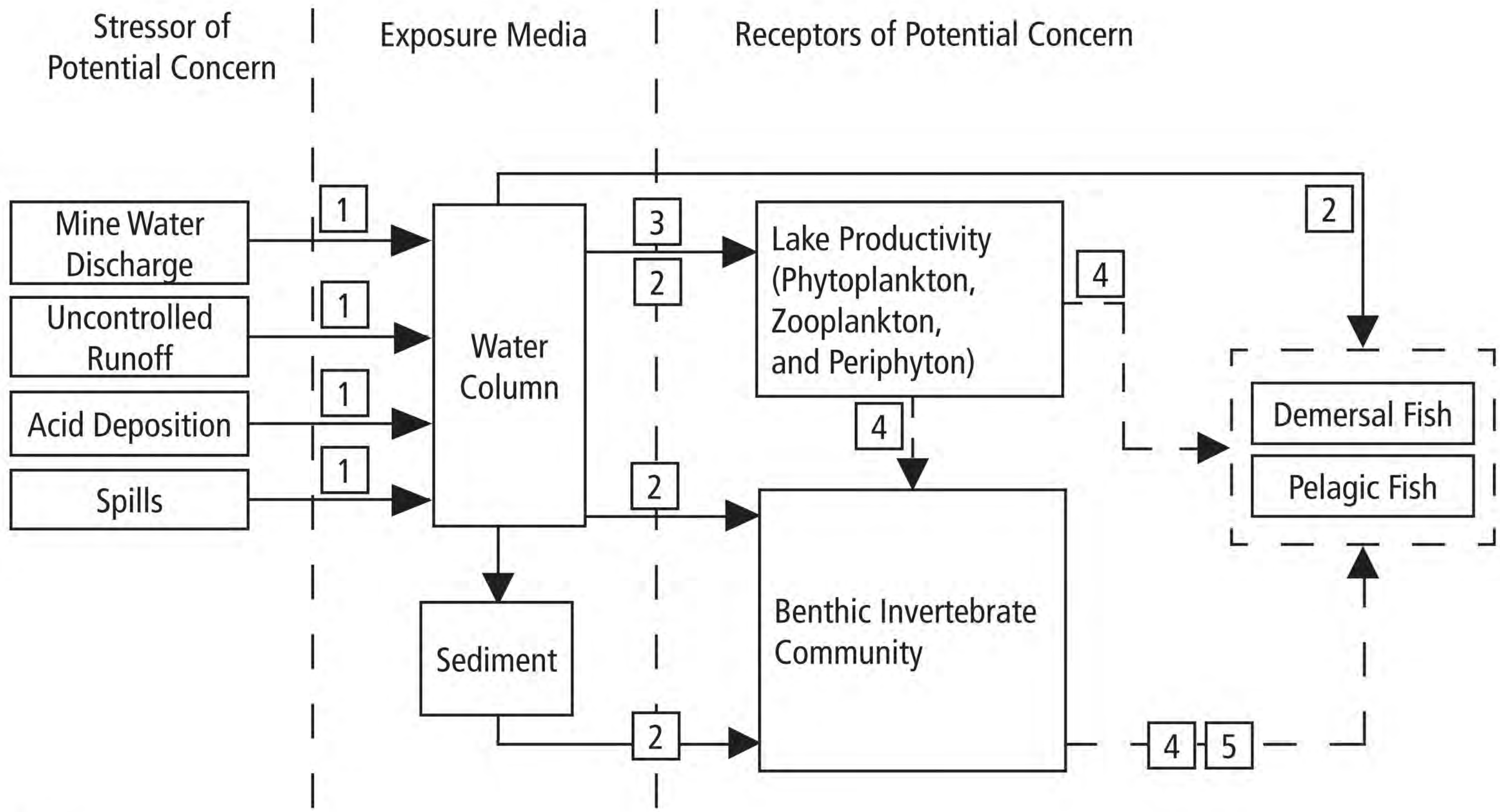
The major exposure pathway relevant to the AEMP is direct contact of aquatic organisms with TDS and associated ions, metals, and nutrients in surface water in Snap Lake (Figure 13-1). Depending on the receptor and the relative concentrations of different chemical stressors, different types of effects may occur in Snap Lake. Periphyton, phytoplankton, and zooplankton are directly exposed to the water column and may be affected by direct toxic effects of TDS and its constituent ions and metals or, in the case of algae, by the growth-stimulating effect of nutrients (N and P) and micronutrients (some components of TDS).

Potential effects of increased concentrations of TDS and its constituent ions, and metals in lake water or sediments, would be largely negative. Zooplankton provide a food supply for pelagic fish, particularly younger life stages and, therefore, any degradation of the zooplankton community resulting from a decreased algal food supply could have a potential indirect effect on the fish community. The benthic invertebrate community is indirectly exposed to sediment porewater, and may be directly exposed to the water column during epibenthic grazing on the sediment surface. The benthic invertebrate community provides a key food supply for demersal and pelagic fish and, therefore, any degradation of the benthic invertebrate community could have a potential indirect effect on the fish community. Demersal and pelagic fish are directly exposed to the water column and may be affected by direct toxic effects from TDS and its constituent ions.

Increased supply of nutrients resulting in enhanced algal growth in the phytoplankton communities would provide an increased food supply to zooplankton, which in turn would result in increased food for fish species or life stages that feed on zooplankton. In addition, enhanced periphyton growth and increased settling rate of organic detritus on the lake bottom from enhanced phytoplankton, periphyton, and zooplankton biomass would provide more food for benthic invertebrates, and ultimately for fish.

Altered balance of nutrients (e.g., increased N, but not P) could affect the aquatic food web through changes in algal biomass, and edibility. A substantial change in the N to P molar ratio can cause phytoplankton community shifts. This in turn can result in a change in food quantity available for zooplankton, because algae in different major groups differ in their degree of edibility or palatability for zooplankton. A decline in zooplankton edibility may result from an increased proportion of inedible or unpalatable algal taxa resulting from an altered balance in nutrients, thereby resulting in decreased zooplankton biomass, and a subsequent decline in the availability of food for fish. Conversely, an altered balance of nutrients may also stimulate the growth of edible algal species, ultimately resulting in an increased quantity of food for fish.

L:\2013\1349\13-1349-0001\Phase 1100\Report A\ Drawing file: 13134800011100A013-1.dwg Apr 23, 2013 - 2:51pm



Notes:

1. Loading of TDS, nutrients, metals and acidifying substances
2. Potential direct toxicity or physical habitat change (sedimentation and dissolved oxygen)
3. Potential nutrient enrichment, or altered balance of nutrients
4. Potential indirect effects due to change in food supply
5. Potential change in tissue chemistry

PROJECT

DE BEERS
GROUP OF COMPANIES

TITLE

CONCEPTUAL SITE MODEL
RELEVANT EXPOSURE PATHWAYS

Golder Associates

PROJECT No. 13-1348-0001.1100			FILE No. 143134900011100A13-1	
DESIGN	TD	2013-02-27	SCALE AS SHOWN	REV. 0
CADD	JEF	2013-04-12	**FIGURE: 13-1**	
CHECK	LMB	2013-04-19		
REVIEW	JEF	2013-04-19		

The preceding discussion describes how inputs of nutrients, metals, and major ions to Snap Lake could result in enrichment, and/or toxicity with the potential to cause impairment of the biological communities in Snap Lake. These pathways can be summarized into two overall hypotheses on the potential effects to Snap Lake from treated effluent release:

- ***Toxicological Impairment Hypothesis*:** Toxicity to aquatic organisms could occur due to substances of toxicological concern (primarily metals, major ions and TDS) released to Snap Lake.
- ***Nutrient Enrichment Hypothesis*:** Eutrophication could occur due to the release of nutrients (primarily phosphorus and nitrogen, and, for some species, TDS and major ions) to Snap Lake.

This qualitative integration provides a systematic approach for distinguishing between these two hypotheses. It is anticipated that these would be the main two types of effects resulting from treated effluent release, but that other naturally-occurring or disturbance-related effects could also occur. These other types of effects are not the focus of the qualitative integration; however, the individual components, and qualitative integration attempt to distinguish these other effects from Mine-related toxicological impairment or nutrient enrichment.

Note that the term “effect” is used in this section in a generic sense to indicate a change (positive or negative) in Snap Lake related to the Mine or Mine activities. It is not intended to reflect the ecological significance or level of concern associated with a given change, nor is it intended to indicate that “pollution[12]” of Snap Lake has occurred.

13.1.2 Endpoints

The 2012 AEMP included parameters and testing representing the following types of information: water quality, and chronic toxicity at the edge of the effluent mixing zone (nutrients and chemical contaminants); sediment quality; fish tissue chemistry; plankton community; benthic invertebrate community; and, fish health. The parameters, and biological variables measured in these components were formulated into endpoints consistent with the key questions addressed by each component section. The types of information provided by the endpoints can be categorized into two endpoint groups representing similar types of evidence:

- ***Exposure:*** Measures of the potential exposure of receptors to Mine-related chemicals and nutrients, including surface water and sediment. In the nutrient enrichment integration, this category also includes indicators of food supply for mid- and upper trophic levels (e.g., for fish, zooplankton biomass, and benthic invertebrate biomass).

[12] The term “pollution” is used to indicate contamination that results in adverse biological effects to populations or communities of organisms.

- ***Field Biological Responses:*** Observationally-based measures of potential ecological changes in the Snap Lake ecosystem, including measures of plankton biomass and community structure, benthic invertebrate abundance and community structure, and fish health.

Quantitative data analysis occurs primarily for the individual AEMP components, and includes individual endpoints that are specific to a particular measurement of the status of the ecosystem. For many of the endpoint groups, multiple endpoints are measured in the AEMP that encompass different stressor types, media, levels of biological organization, and data analysis methods, providing a "battery" approach for assessing the degree of effect associated with each group.

13.1.3 Endpoint Response Ratings

The starting point for the qualitative integration is rating of the endpoint results from each component according to a series of decision criteria. These endpoint ratings then "feed into" the analysis, where weighting considerations are applied qualitatively (Section 13.1.4), and then combined to obtain the overall conclusion.

The observed changes, differences, trends, and/or exceedances of benchmarks in exposure, and field biological response endpoints, are classified using semi-quantitative descriptions of the responses or degree of changes observed in Snap Lake. The list of response ratings for the 2012 AEMP is presented in Table 13-1. Increasingly large and/or statistically significant responses in Snap Lake receive progressive ratings of "No response" (represented by 0), "Rating 1" (represented by "↑" or "↓"), "Rating 2" (represented by "↑↑" or "↓↓"), or "Rating 3" (represented by "↑↑↑" or "↓↓↓") depending on the magnitude and direction of the response. The arrows provide a visual description of the direction of response (e.g., ↑ = increase, ↓ = decrease); up and down arrows are combined for endpoints where the direction of response is not as apparent, such as metrics of community structure. Narrative descriptions of the ratings are provided below:

- **No Response –** Typically, a finding of no exceedance of a prediction or benchmark, no visual and/or statistical difference, no trend, or no difference in trend (Snap Lake versus reference) will indicate a rating of "no response".
- **Rating 1 –** This rating indicates that a change, response, or trend in exposure may be apparent in Snap Lake, or that a conservative numerical benchmark has been exceeded, but that the linkage to broader ecosystem effects is weak and changes are reversible. It also includes indications of minor shifts (i.e., at the species or genus level) in the abundance, richness, or community structure of the phytoplankton, zooplankton, or benthic communities, as well as, minor changes/trends in fish population and health indicators.

Table 13-1 Preliminary Response Ratings for the Weight of Evidence Assessment

Endpoint Group	Endpoint	No Response	Rating 1 ↑/↓[(a)]	Rating 2 ↑↑/↓↓[(a)]	Rating 3 ↑↑↑/↓↓↓[(a)]
Exposure – Water Quality (potential toxicants and measured mixing zone toxicity)	**Comparison to Benchmarks** (where they exist)	<EAR Prediction	>AEMP Benchmark[(b)]	>Site-specific guideline[(c)]	Rating 2 in at least two endpoints. **OR** Persistent lethal toxicity
	Trends Snap Lake compared to reference lakes	No difference	Trend difference between Snap Lake and reference	Trend difference outside confidence interval (if applicable)	
	Comparison to baseline normal range	No difference	Difference in mean concentration	Snap Lake mean >baseline normal range[(d)]	
	Toxicity at edge of mixing zone	No persistent toxicity	Sublethal toxicity observed at edge of mixing zone in 2 or more consecutive monitoring events	Persistent sublethal toxicity with trend to increasing in frequency or severity	
Exposure – Water Quality (nutrients)	**Comparison to AEMP Benchmarks** (where they exist)	<EAR Prediction	>AEMP Benchmark	>Site-specific guideline	Rating 2 in at least two endpoints. **OR** Rating 1 in a downstream lake
	Trends Snap Lake compared to reference lakes	No difference	Trend difference between Snap Lake and reference	Trend difference outside confidence interval (if applicable)	
	Comparison to baseline normal range	No difference	Difference in mean concentration	Snap Lake mean >baseline normal range	
Exposure – Sediment Quality (potential toxicants)	**Comparison to Benchmarks** (where they exist)	<ISQG	>ISQG	>PEL	Rating 2 in at least two endpoints.
	Snap Lake compared to reference lakes and baseline normal range	No difference	Statistically significant increase in Snap Lake	Statistically significant increase beyond normal range	
	Temporal Trends	No trend	Statistically significant increasing trend in Snap Lake	Statistically significant increasing trend[(e)] in Snap Lake, at a magnitude of toxicological concern[(f)].	

Table 13-1 Preliminary Response Ratings for the Weight of Evidence Assessment

Endpoint Group	Endpoint	No Response	Rating 1 ↑/↓[a]	Rating 2 ↑↑/↓↓[a]	Rating 3 ↑↑↑/↓↓↓[a]
Exposure – Fish Tissue Chemistry (potential toxicants)	**Snap Lake compared to reference lakes**	No difference	Difference in mean concentration	Snap Lake mean >normal range	Rating 2 in both endpoints
	Snap Lake compared to baseline	No difference	Difference in mean concentration	Snap Lake mean >normal range	
Field Biological Responses – Plankton Community	**Trends Snap Lake compared to reference lakes** Chlorophyll *a*, Phytoplankton Abundance/Biomass, Zooplankton Abundance/Biomass	No trend difference	Trend difference between Snap Lake and reference	Trend difference outside confidence interval (if applicable)	Rating 2 in at least two endpoints
	Snap Lake compared to Baseline (i.e., 2004) Phytoplankton Abundance/Biomass, Zooplankton Abundance/Biomass	No difference	Difference (mean vs mean) outside the normal range	Exceeding EAR predictions	
	Community Structure Phytoplankton and Zooplankton Communities	No difference	Minor shift in community structure (i.e., at species/genus level)	Moderate shift in community structure (i.e., at class or functional group level)	

Table 13-1 Preliminary Response Ratings for the Weight of Evidence Assessment

Endpoint Group	Endpoint	No Response	Rating 1 ↑/↓[a]	Rating 2 ↑↑/↓↓[a]	Rating 3 ↑↑↑/↓↓↓[a]
Field Biological Responses – Benthic Community	**Trends Snap Lake compared to reference lakes** Density, Richness, Densities of Dominant Taxa, Community Structure Variable	No difference	Trend difference between Snap Lake and reference	Trend difference outside confidence interval (if applicable)	Rating 2 in at least two endpoints
	Snap Lake compared to reference lakes Density, Richness, Densities of Dominant Taxa, Community Structure Variable	No difference	Statistical difference	Statistical difference beyond normal range	
	Community Structure Benthic Community	No change	Minor shift in community structure (i.e., at genus level)	Moderate shift in community structure (i.e., at major group level)	
Field Biological Responses – Fish Health and Community	**Fish Health** Survival, Growth, Reproduction and Condition Endpoints	No difference	Statistical difference	Statistical difference beyond normal range	To be developed
	Fish Community Endpoints to be developed	No difference	To be developed	To be developed	To be developed

Notes:

(a) The direction of the arrow, up or down, indicates the direction of change or relationship (i.e., increase/positive versus decrease/negative). For biological community structure endpoints, both arrows are included (↑/↓) to reflect that a community shift normally involves combined increases and decrease in abundance and diversity. ↑/↓ = Rating 1; ↑↑/↓↓ = Rating 2; ↑↑↑/↓↓↓ = Rating 3.

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

(c) Site-specific benchmarks for Snap Lake that may be developed under the AEMP Response Framework.

(d) “Normal Range” is determined based on +/- 2SD in Snap Lake main basin baseline and +/- 2SD in reference lakes, and/or other appropriate considerations.

(e) Note that this Rating criterion is hypothetical at this stage because statistical methods for trend analysis have yet to be established.

(f) To be determined on a substance-by-substance basis considering proximity to or exceedance of benchmarks and the normal range.

AEMP = Aquatic Effects monitoring Program; EAR = Environmental Assessment Report; EA = Environmental Assessment; ISQG = Interim Sediment Quality Guideline; PEL = Probable Effect Level; SD = Standard Deviation.

- **Rating 2 –** This rating includes situations where greater changes, responses, or trends in exposure (i.e., outside normal range[13]), and exceedances of less conservative numerical values such as generic water quality or sediment quality guidelines have occurred, and the changes appear to be linked to the Mine. It also includes indications of moderate shifts (i.e., at the class or functional group level) in the abundance, richness, or community structure of the phytoplankton, zooplankton, or benthic communities, as well as, marked changes or trends in fish population and health indicators.
- **Rating 3 –** This rating indicates the strongest level response in exposure or biological response endpoints. None of the endpoints in the qualitative integration conducted in the 2012 AEMP were judged to be at this rating. It is anticipated that this rating would be applied when multiple endpoints within a group are found to be at Rating 2, indicating a strong level of evidence for response for a given indicator of exposure (water quality, sediment quality, or fish tissue chemistry) or biological response (plankton community, benthic community, or fish community/health). As additional years of AEMP data are obtained and the WOE Approach is refined, the conditions under which this rating is applied will be developed further and refined.

For each endpoint group, the highest observed level of response was typically carried through the analysis, since these highest responses provide the early-warning indicator of potential adverse effects to the Snap Lake ecosystem. In cases where the highest level response was not considered representative, a rationale was provided for why other endpoints were considered more representative.

Application of the ratings typically erred on the side of caution (i.e., in the direction of a false-positive) to represent the potential worst-case responses in the component endpoints. This meant that, when a rating was apparently achieved, then it was typically applied even if the degree of trend or change was mild, or if there was uncertainty in the finding, or potential alternative causes of the endpoint response.

13.1.4 Weighting Considerations

Weighting was applied qualitatively in the framework, and included *a priori* considerations that were independent of the actual AEMP findings, consideration of the direction of change or response, and *a posteriori* considerations based on the nature, complexity, and uncertainty of the AEMP findings.

A priori considerations were based on professional judgement regarding the strength and relevance of the evidence contributed by a particular endpoint and were applied to an endpoint regardless of the endpoint result. The overall purpose of *a priori* weighting is to capture representativeness, or the “ability” of an endpoint to indicate actual responses in Snap Lake. Actual biological responses in Snap Lake are deemed to provide a more direct indicator of

[13] “Normal Range” is determined based on +/- 2SD in Snap Lake Main Basin baseline and +/- 2SD in reference lakes, and/or other appropriate considerations.

potential effects in the aquatic ecosystem than indicators of exposure to nutrients and chemicals, or laboratory toxicity testing, and will therefore have higher *a priori* weighting. Exposure indicators do not consider the dose-response relationship between exposure and response, or factors that affect bioavailability under natural conditions. Laboratory cultures used in toxicity testing are often more sensitive than typically more tolerant natural populations, meaning that responses observed in the laboratory may not occur or be as pronounced in natural systems. Higher weighting for field biological response endpoints is consistent with guidance from the literature that field-based effect studies should be weighted higher than laboratory and chemistry-based analyses (Chapman and Anderson 2005; Wenning et al. 2005; Environment Canada and Ontario Ministry of the Environment 2008; Chapman and Smith 2012).

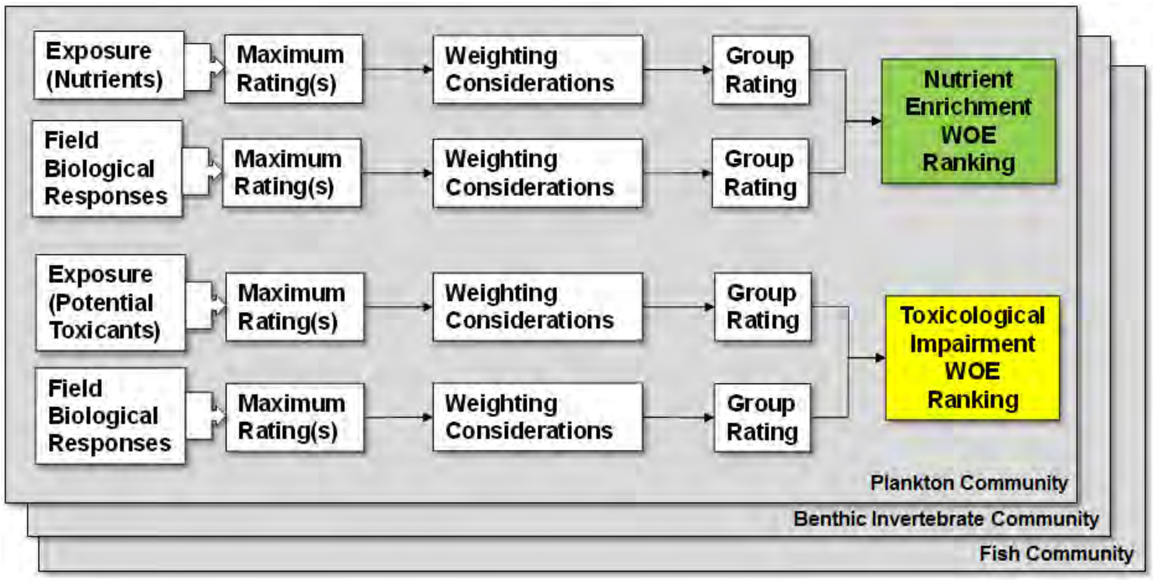
Direction considerations were applied to field biological response endpoints only to reflect the degree of support that an observed biological response contributes to the two alternative hypotheses. These considerations were contingent on the observed direction of change or relationship. For example, increases in plankton biomass would typically only be expected as a result of nutrient enrichment, and therefore provide 100 percent (%) support for this hypothesis. Conversely, changes in plankton community structure might be expected as a result of either nutrient enrichment or toxicological impairment, and therefore provide proportional support for each hypothesis but at a level less than 100%. In some cases, responses were observed for a particular endpoint which was opposite to those which would be expected for a given hypothesis. Where this information was considered important to the integration process, the response and direction (i.e., as indicated by up or down arrows [↑ or↓]), was included, but the arrow was put in brackets to indicate that the particular response did not support the hypothesis being examined. The hypothesis supported by a given biological response is discussed further in the endpoints summaries for each AEMP component in Section 13.2.1.

A posteriori considerations were applied where appropriate to reflect additional insight gained during data collection, and analysis. Thus, this consideration reflected best professional judgement regarding the AEMP findings for 2012. Two relevant factors are consistency in response among the individual endpoints within an endpoint group, and strength of linkage to treated effluent release (for exposure endpoints) and exposure (for biological response endpoints). Where *a posteriori* weighting was applied in the qualitative integration, a discussion of the rationale was provided.

13.1.5 Integration

The final step is integration of the results of endpoints for exposure and field biological responses to provide a qualitative determination of the level of support for each hypothesis (nutrient enrichment versus toxicological impairment), separated by ecosystem component (plankton community, benthic invertebrate community, and fish community). Figure 13-2 provides a graphical summary of the overall integration process.

Figure 13-2 Conceptual Integration Process Applied in the Weight of Evidence Assessment



For each component, the outcome of the qualitative determination resulted in a WOE Ranking that indicates the strength of support for each of the two alternative hypotheses according to the following scheme:

- **WOE Rank 0** – Hypothesis not supported by the combined endpoint findings;
- **WOE Rank 1** – Hypothesis has weak support from the combined endpoint findings;
- **WOE Rank 2** – Hypothesis has moderate support from the combined endpoint findings; and,
- **WOE Rank 3** – Hypothesis has strong support from the combined endpoint findings.

The rankings are intended to reflect the analyses in the component reports, and response ratings specific to each endpoint. In particular, they provide an indication of the relative strength of evidence associated with apparent Mine-related changes, responses, or effects by a particular ecosystem component. A higher rank represents a higher strength of support for a particular hypothesis. The integration process includes a side-by-side comparison of exposure and biological response endpoints, along with documentation of how weighting and judgement have been applied with the purpose of providing transparency in the integration process.

An important consideration is that the WOE Rankings are not intended to indicate the ecological significance of observed effects. For example, it is possible that there could be moderate evidence (WOE Rank 2) for a particular hypothesis in Snap Lake, but that the magnitude and significance with respect to the ecological integrity of Snap Lake could be relatively mild. This is an important distinction between the qualitative integration and the AEMP Response Framework described in the Draft AEMP Design Plan. The qualitative integration describes potential linkages from exposure to observed biological differences and changes in Snap Lake, and actively supports decision-making in the AEMP Response Framework, which sets specific levels of acceptable or unacceptable effects with respect to the ecological integrity of Snap Lake, on a component-by-component basis.

13.2 RESULTS

13.2.1 Endpoint Summaries

Tables 13-2 to 13-8 provide the endpoint summaries for each AEMP component. The endpoint summaries categorize the responses for the endpoints associated with each AEMP component according to the response ratings presented in Table 13-1. Further discussion of the endpoint findings for each component is provided in the component chapters (Sections 3 through 7, and Section 9).

For ease of interpretation and presentation of the summaries for water and sediment quality, the parameters were grouped into two overall categories: (i) parameters with benchmarks; and, (ii) parameters without benchmarks, with a further distinction between toxicological benchmarks and enrichment benchmarks.

Table 13-2 Water Quality Endpoint Summary

Endpoint Group	Parameter Grouping	List of Parameters	Endpoint Ratings: Comparison to Benchmarks	Endpoint Ratings: Trends in Snap Lake Compared to Northeast Lake	Endpoint Ratings: Comparison of Snap Lake Main Basin to Normal Range
Toxicants					
Parameters Without Toxicological Benchmarks	Major Ions/Constituents of TDS	calcium, magnesium, sodium, sulphate, potassium,	n/a	↑	↑↑
	Possible Toxicants	barium, rubidium, strontium, lithium	n/a	↑	↑↑
	Non-responsive Parameters	manganese, titanium, phosphorus, turbidity, antimony	n/a	no response	no response
Parameters with Toxicological Benchmarks	Major Ions	chloride, fluoride, nitrate	↑	↑	↑↑
	TDS	calculated TDS	no response	↑	↑↑
	Possible Toxicants That Were Below Benchmarks	nitrite, ammonia, boron[(a)], molybdenum[(a)], uranium[(a)]	no response	↑	↑↑
		arsenic[(a)], nickel[(a)]	no response	↑	no response
		zinc	no response	no response	↑
		cadmium, hexavalent chromium	no response	no response	no response
	pH	laboratory pH	no response	no response	↑↑
	Non-responsive Parameters	aluminum, copper, iron, lead, mercury, selenium, silver, thallium, total chromium	no response	no response	no response
Nutrients					
Parameters Without Enrichment Benchmarks	Nitrogen compounds	nitrate, nitrite, ammonia	n/a	↑	↑↑
	TKN	TKN	n/a	↑	no response
Parameters With Enrichment Benchmarks	TDS	Calculated TDS	no response	↑	↑↑
	Phosphorus	Dissolved and total phosphorous	no response	no response	no response

Note: The direction of the arrow, up or down, indicates the direction of change or relationship (i.e., increase/positive versus decrease/negative). ↑/↓ = Rating 1; ↑↑/↓↓ = Rating 2.

(a) Parameter concentration is well-below the benchmark, suggesting that the trends and differences from normal range are of low toxicological significance.

n/a = not applicable for this parameter grouping; TDS = total dissolved solids; TKN = total Kjeldahl nitrogen

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Table 13-3 Mixing Zone Toxicity Endpoint Summary

Endpoint Group	Endpoint	List of Parameters	Toxicity at Edge of Mixing Zone
Laboratory Toxicity	Algae Toxicity	*Pseudokirchneriella subcapitata* growth	no response
	Invertebrate Toxicity	*Ceriodaphnia dubia* survival	no response
		Ceriodaphnia dubia fecundity	no response

Table 13-4 Sediment Quality Endpoint Summary

Endpoint Group	List of Parameters	Endpoint Ratings		
		Comparison to Benchmarks	Comparison to Reference and Baseline Normal Range	Temporal Trends in Snap Lake Main Basin
Toxicants				
Parameters Without Benchmarks	Calcium	n/a	↑	no response
	antimony, tin[(a)]	n/a	↑↑	no response
	bismuth, selenium, sodium, strontium	n/a	↑↑	↑
	remaining parameters	n/a	no response	no response
Parameters with Benchmarks	cadmium, chromium, copper, zinc	↑	no response	no response
	arsenic, lead, mercury	no response	no response	no response
Nutrients				
Parameters Without Benchmarks	TOC, available ammonium	n/a	no response	no response
	available nitrate, TKN, total nitrogen	n/a	↑	no response
	available phosphate	n/a	no response	no response
	available potassium, available sulphate	n/a	no response	↑

Note: The direction of the arrow, up or down, indicates the direction of change or relationship (i.e., increase/positive versus decrease/negative). ↑/↓ = Rating 1; ↑↑/↓↓ = Rating 2.

(a) Detected for first time in 2012.

n/a = not applicable for this parameter grouping; TOC = total organic carbon; TKN = total Kjeldahl nitrogen

Table 13-5 Fish Tissue Chemistry Endpoint Summary

Endpoint Group	List of Parameters	Endpoint Ratings	
		Snap Lake Compared to Reference Lakes	Snap Lake Compared to Baseline Normal Range
Fish Tissue Chemistry	strontium	↑	n/a
	thallium	↑↑	n/a
	remaining parameters	no response	n/a

Note: The direction of the arrow, up or down, indicates the direction of change or relationship (i.e., increase/positive versus decrease/negative).
↑/↓ = Rating 1; ↑↑/↓↓ = Rating 2.
n/a = not conducted for 2012 because baseline data are not available for Lake Chub.

Table 13-6 Plankton Community Endpoint Summary

Endpoint Group	Endpoint	Rating	Description	Hypothesis Supported
Phytoplankton				
Chlorophyll *a*	Trends in Snap Lake compared to reference lakes	no response	-	-
	Snap Lake compared to baseline	no response	-	-
Phytoplankton Biomass	Trends in Snap Lake compared to reference lakes	↑ and ↓	There was a clear increasing trend from 2004 to 2009, followed by a decreasing trend back to near baseline.	Enrichment
	Snap Lake compared to baseline	no response	-	-
Community	Community Structure	↑↑/↓↓	Changes in relative biomass/abundance at functional group level (shift from chrysophyceae-cyanobacteria dominated community to diatom dominated)	Enrichment
Zooplankton				
Zooplankton Biomass	Trends in Snap Lake compared to reference lakes	no difference	-	-
	Snap Lake compared to baseline	↓	Decrease in Snap Lake Compared to Baseline	Toxicity
Community	Community Structure	↑/↓	Minor community shift observed in NMDS plots	Either

Note: The direction of the arrow, up or down, indicates the direction of change or relationship (i.e., increase/positive versus decrease/negative). For biological community structure endpoints, both arrows are included (↑/↓) to reflect that a community shift normally involves combined increases and decrease in abundance and diversity.
↑/↓ = Rating 1; ↑↑/↓↓ = Rating 2.
NMDS = non-metric multidimensional scaling; “-“= information not provided for non-responsive endpoints.

Table 13-7 Benthic Invertebrate Community Endpoint Summary

Endpoint Group	Endpoint	Rating	Description	Hypothesis Supported
Total Density	Trends in Snap Lake compared to Northeast Lake	↓	Slight decreasing trend in Snap Lake compared to Northeast Lake	Toxicity
	Snap Lake compared to Northeast Lake	no difference	-	-
Richness	Trends in Snap Lake compared to Northeast Lake	no difference	-	-
	Snap Lake compared to Northeast Lake	↓	Lower richness in Snap Lake than Northeast Lake	Toxicity
Diversity	Trends in Snap Lake compared to Northeast Lake	no difference	-	-
	Snap Lake compared to Northeast Lake	no difference	-	-
Evenness	Trends in Snap Lake compared to Northeast Lake	↑	Decreasing evenness trend in Northeast Lake but evenness relatively constant in Snap Lake	Either
	Snap Lake compared to Northeast Lake	no difference	-	-
Density of Dominant Taxa	Trends in Snap Lake compared to Northeast Lake	↓ (*Microtendipes* and Pisidiidae)	Decreasing trend in Snap Lake compared to Northeast Lake, for *Microtendipes* and Pisidiidae	Toxicity
	Snap Lake compared to Northeast Lake	↑ (*Valvata* only)	*Valvata* density is higher in Snap Lake relative to Northeast Lake	Either
Community	Community Structure	no difference	-	-

Note: The direction of the arrow, up or down, indicates the direction of change or relationship (i.e., increase/positive versus decrease/negative). For biological community structure endpoints, both arrows are included (↑/↓) to reflect that a community shift normally involves combined increases and decrease in abundance and diversity.
↑/↓ = Rating 1.
"-"= information not provided for non-responsive endpoints

Table 13-8 Fish Health Endpoint Summary

Endpoint Group	Endpoint	Comparison to Pooled Reference Lakes			Description	Hypothesis Supported
		Male	Female	Juvenile		
Survival	Age	↓	↓	no response	All size classes present in Snap Lake compared to the reference lakes	Neither
	Length-Frequency Distribution	↑/↓			Differences indicate varying proportions among the lakes but are not indicative of a Mine-related effect	Neither
Growth (Energy Use)	Length	↓	↓	↑	Smaller adults but larger juveniles in Snap Lake	Toxicity (m/f) *and* Enrichment (j)
	Weight	↓	↓	↑	Smaller adults but larger juveniles in Snap Lake	Toxicity (m/f) *and* Enrichment (j)
	Size at Age 1-yr			↑	Larger juveniles in Snap Lake	Enrichment
	Size at Age 2-yr	↓	no response		Smaller males in Snap Lake	Toxicity(m)
	Size at Age 3-yr	no response	↑		Larger females in Snap Lake	Enrichment (f)
Reproduction (Energy Use)	GSI	↑/no response	no response		Greater male GSI based on ANOVA but not ANCOVA	-
	Egg diameter		↓		Smaller gonad weight and egg diameter in adult females	Toxicity
	Fecundity		no response		-	-
	Relative fecundity		↑		Higher relative fecundity in adult females	Enrichment
Condition (Energy Storage)	K	no response	no response	no response	-	-
	LSI	no response	no response	no response	-	-

Note: Shaded cells indicate that this endpoint is not applicable to the life-stage or sex.

The direction of the arrow, up or down, indicates the direction of change or relationship (i.e., increase/positive versus decrease/negative). For biological community structure endpoints, both arrows are included (↑/↓) to reflect that a community shift normally involves combined increases and decrease in abundance and diversity. ↑/↓ = Rating 1.

m = male; f = female; j = juvenile; ANOVA = analysis of variance; ANCOVA = analysis of covariance; GSI = gonado-somatic index; K = condition factor; LSI = liver-somatic index; "-"= information not provided for non-responsive endpoints.

Within each of these categories, subsets of parameters were grouped based on consistency in response with respect to comparison to benchmarks (applied only for the first category), trends, and differences from the normal range. These groupings and response ratings were conducted separately for parameters typically expected to be potential aquatic toxicants, and for those typically expected to be nutrients. Note that some parameters, such as TDS and nitrate can act as both toxicants and nutrients, and were therefore included in the groupings for both types of responses.

For the biological response endpoints, including those for the plankton community, the benthic invertebrate community, and fish health, the endpoints were grouped by the biological variable being examined with ratings assigned to multiple endpoints within each biological variable (i.e., trends, differences from reference or normal range, and community structure). Note that where Snap Lake versus reference comparisons were made, the "reference" lakes included Northeast Lake only for plankton and benthos but included both Northeast Lake and Lake 13 for fish tissue and fish health. Note further that fish tissue chemistry could only be assessed for potential toxicant exposure, not nutrient exposure, because nutrients were not analyzed in fish tissue.

For each endpoint where a response was observed in 2012, a preliminary judgement was made regarding which hypothesis the response supported. These judgements presume that nutrient enrichment or toxicological impairment are the only factors acting on endpoints in Snap Lake (i.e., they answer the question: *If nutrient enrichment or toxicological impairment are the only factors acting on endpoints, which of the two hypotheses would this type of response typically support?*). These judgements were used to support direction weighting considerations in the qualitative integration and answers included *toxicity* (toxicological impairment hypothesis), *enrichment* (nutrient enrichment hypothesis), *either* (i.e., where the change could support both hypotheses), or *neither* (i.e., where there was clearly an alternative explanation for the observed changes).

For plankton and invertebrates, an increase or positive trend in community biomass indicators (total density or biomass, dominant species density, and chlorophyll *a*), or richness with treated effluent exposure, typically provides a high level of support for nutrient enrichment. In the absence of other factors, these types of responses would usually only be expected to result from nutrient enrichment. For biomass indicators, the converse is also true, with a decrease or negative trend providing a high level of support for toxicological impairment. However, a decrease in richness could possibly result from toxicological impairment (i.e., selective toxicity) or nutrient enrichment (i.e., one dominant species out-competing other species). Also, densities of individual species might respond counter to these generalizations in situations where toxicological impairment reduced competition for a tolerant species.

Multiple indicators of community structure, such as diversity, evenness, relative abundance, are typically equivocal with respect to the degree of support for each hypothesis. These endpoints can indicate a change or trend relative to a reference area or baseline condition; however, the cause of a change in the biological community is less clear and may depend on the responses of other variables. This uncertainty notwithstanding, the inclusion of these types of endpoints is important because changes in community structure can often be more sensitive than the biomass or richness responses, making community structure an early warning of change that should be further investigated.

Specific considerations applied for the plankton and benthic invertebrate responses were:

- The trend in phytoplankton biomass has been equivocal, with an increasing trend prior to 2009 but then a decreasing trend. However, phytoplankton biomass remained higher in Snap Lake than in Northeast Lake, and on balance the pattern appears to support an explanation of enrichment, and then a compensatory shift in community structure (i.e., to a diatom dominated community).
- The decreased biomass of zooplankton in Snap Lake compared with a minor community shift appeared consistent with toxicological impairment, in the absence of other influences such as food supply, predation, or inter-annual variation in regional factors (e.g., temperature, light). These changes were considered mild, especially given that zooplankton biomass in Snap Lake was still greater than in Northeast Lake.
- For benthic invertebrates, the decreasing trend in total density, lower richness in Snap Lake, and decreasing trends for *Microtendipes* and Pisidiidae, all appeared to support toxicity in the absence of other influences such as food supply, predation, or inter-annual variation. All responses were considered mild and consistent with EAR predictions.
- The hypothesis supported by the higher *Valvata* density in Snap Lake compared to Northeast Lake was considered uncertain because the difference was relatively small, and the increase in *Valvata* could be due to selective enrichment of this species or could be due to reduced competition due to the decreasing trends in other dominant species.
- The differing trends with respect to evenness were not deemed to discriminate between the two hypotheses because evenness remains relatively constant in Snap Lake, but is decreasing in Northeast Lake.

For fish, increases between Snap Lake and reference lakes and/or increasing trends in growth, reproductive investment, and condition parameters suggest improved fish health, which would typically only be expected in response to enrichment of food supply and resources. Conversely, decreases in these parameters may indicate diminished fish health and suggest the possibility of toxicological impairment. Alternative explanations for decreases could also be decreased food supply or increased predation, which results in a re-allocation of resources away from maintenance, growth, and reproduction.

Specific considerations applied for the 2012 fish health responses were:

- With respect to growth parameters, results were mixed, with juveniles and 3-year (yr) adult females typically indicating increased growth relative to reference lakes, but 2-yr adult males indicating decreased growth. Pooled adults in the non-lethal survey indicated decreases in growth relative to the reference lakes, based on length and weight. Both directions of response for the age classes and pooled adults were included in the qualitative integration.

- Some reproductive parameters increased (i.e., relative fecundity) while others decreased (egg size); both directions of response were included in the qualitative integration.
- As described in Section 7.3.2, otolith age data were inconsistent with maximum Lake Chub ages reported in the literature, and also inconsistent with fish health endpoints included in the present study (i.e., fish length, weight and state of maturity as indicated by gonad histology were inconsistent with otolith age). As such, length frequency distributions were used to assign age but uncertainty in the assigned ages meant that limited interpretation and statistical analyses were performed with respect to age endpoints, for juvenile fish in particular.

Additional discussion of the endpoint responses relevant to each hypothesis is provided in the analysis that follows.

13.2.2 Toxicological Impairment Analysis

The qualitative integration describing the integration for potential toxicological impairment of the Plankton Community, the Benthic Invertebrate Community, and the Fish Community is summarized in Table 13-9.

13.2.2.1 Plankton Community

The endpoint findings and rationale for the rating of each endpoint group for The Plankton Community are as follows:

- ***Exposure:*** Water quality in Snap Lake is the main indicator of exposure for the plankton community. For 2012 the water quality parameters that exhibited the strongest and most consistent responses in Snap Lake were chloride, fluoride, and nitrate, as well as TDS and its constituent ions. Chloride, fluoride, and nitrate each had exceedances of their respective Canadian Council of Ministers of the Environment (CCME) water quality guidelines, combined with apparent increasing trends in Snap Lake and concentrations that were outside of the baseline normal range. Exceedance of conservative benchmarks for these parameters means that the trends and the Snap Lake mean outside of the baseline normal range could be of toxicological relevance, and further study is likely warranted (i.e., via site-specific guideline development). Total dissolved solids did not exceed the AEMP benchmark in 2012 (the AEMP benchmark was the predicted maximum whole-lake TDS concentration of 350 milligrams per litre [mg/L]), but it along with its constituent ions (calcium, magnesium, sodium, sulphate, potassium, chloride) showed increasing trends in Snap Lake and concentrations that were outside of the baseline normal range. At a lesser response level, some metals without AEMP benchmarks were also noteworthy including barium, lithium, rubidium, and strontium; these metals are found in the treated effluent signature, and were found to have increasing trends in Snap Lake and concentrations that were outside of the baseline normal range. These combined findings for water quality endpoints resulted in a maximum response rating of Rating 2.

However, since the primary source of fluoride, chloride, and nitrate is the treated effluent, increases in these parameters are associated with elevated calcium and hardness, which are expected to reduce the potential for toxicity effects associated with fluoride, chloride, and nitrate. In addition, there was no toxicity to algae and water flea observed at the diffuser mixing zone suggesting a lack of direct water toxicity in short-term chronic exposures. Thus, there is uncertainty as to the actual toxicological significance of the benchmark exceedances, trends, and differences in Snap Lake water quality. Water quality, overall, was judged to be at Rating 1 for this exposure endpoint group.

Table 13-9 Qualitative Integration for the Toxicological Impairment Hypothesis

<table>
<tr><th>Endpoint Group</th><th>Endpoint</th><th>Maximum Response Rating</th><th>Description of Response</th><th>Group Rating</th><th>WOE Rank and Rationale</th></tr>
<tr><td colspan="6">(a) Plankton Community</td></tr>
<tr><td rowspan="4">Water Quality (Exposure)</td><td>Comparison to Benchmarks</td><td>↑</td><td>Chloride, fluoride, nitrate exceed AEMP benchmarks.</td><td rowspan="4">↑</td><td rowspan="9">WOE Rank 1

-WQ exposure has increased for key parameters and is exceeding generic conservative guidelines, but not site-specific benchmarks. There was no indication of treated effluent toxicity at the edge of the mixing zone in 2012.
-Zooplankton response is consistent with mild toxicological impairment, but could also be related to trophic dynamics such as top-down feeding pressure and/or food supply.
-Phytoplankton response does not appear consistent with toxicological impairment; it is better explained by nutrient enrichment and a community shift.</td></tr>
<tr><td>Trends in Snap Lake Compared to Reference Lakes</td><td>↑</td><td>Chloride, fluoride, nitrate, TDS, multiple metals, additional major ions have increasing trends relative to Northeast Lake.</td></tr>
<tr><td>Comparison to Baseline Normal Range</td><td>↑↑</td><td>Concentrations of chloride, fluoride, nitrate, TDS, multiple metals, and additional major ions exceed baseline concentrations.</td></tr>
<tr><td>Toxicity at Edge of Mixing Zone</td><td>no response</td><td>-</td></tr>
<tr><td rowspan="3">Phytoplankton Community (Field Biological Response)</td><td>Chlorophyll a</td><td>no response</td><td>-</td><td rowspan="3">↑/↓</td></tr>
<tr><td>Phytoplankton Biomass</td><td>(↑) and ↓</td><td>Increasing trend from 2004 to 2009, decreasing trend back to near baseline from 2009 to present.</td></tr>
<tr><td>Community Structure</td><td>↑↑/↓↓</td><td>Shift from chrysophyceae-cyanobacteria dominated community to diatom dominated community.</td></tr>
<tr><td rowspan="2">Zooplankton Community (Field Biological Response)</td><td>Zooplankton Biomass</td><td>↓</td><td>Decrease in Snap Lake compared to baseline.</td><td rowspan="2">↓</td></tr>
<tr><td>Community Structure</td><td>↑/↓</td><td>Minor community shift observed in NMDS plots.</td></tr>
</table>

Table 13-9 Qualitative Integration for the Toxicological Impairment Hypothesis

Endpoint Group	Endpoint	Maximum Response Rating	Description of Response	Group Rating	WOE Rank and Rationale
(b) Benthic Invertebrate Community					
Water Quality (Exposure)	Overall Assessment	↑↑	see Plankton Community(above)	↑↑	**WOE Rank 1** -WQ exposure has increased for key parameters and is exceeding generic conservative guidelines but not site-specific benchmarks. There was no indication of treated effluent toxicity at the edge of the mixing zone in 2012. -Multiple sediment metals are displaying increasing temporal trends and are beyond the baseline normal range, for Snap Lake main basin. However, none of the metals that exceeded the ISQG are indicating differences from reference or baseline conditions, or trends in Snap Lake. -The benthic community response is consistent with mild toxicological impairment but could also be caused by inter-annual variation. These results appear different from previous years; continued monitoring will help determine whether this was due to random variability or the beginning of toxicological impairment in Snap Lake.
Sediment Quality (Exposure)	Comparison to Benchmarks	↑	Cadmium, chromium, copper, and zinc exceed the CCME ISQG.		
	Snap Lake Compared to Reference Lakes and Baseline	↑↑	Antimony, bismuth, selenium, sodium, strontium, and tin are higher than in Northeast Lake and beyond the normal range in Snap Lake		
	Temporal Trends	↑	Bismuth, selenium, sodium, strontium and tin are increasing in Snap Lake main basin		
Benthic Invertebrate Community (Field Biological Response)	Total Density	↓	Slight decreasing trend in Snap Lake compared to Northeast Lake.	↓	
	Richness	↓	Lower richness in Snap Lake than Northeast Lake in 2012.		
	Diversity	no response	-		
	Evenness	↑	Decreasing evenness trend in Northeast Lake but evenness remains relatively constant in Snap Lake.		
	Density of Dominant Taxa	↓	Decreasing trend in Snap Lake compared to Northeast Lake, for *Microtendipes* and Pisidiidae		
		(↑)	*Valvata* density is higher in Snap Lake relative to Northeast Lake		
	Community Structure	no difference	-		

Table 13-9 Qualitative Integration for the Toxicological Impairment Hypothesis

Endpoint Group	Endpoint	Maximum Response Rating	Description of Response	Group Rating	WOE Rank and Rationale
(c) Fish Community					
Water Quality (Exposure)	Overall Assessment	↑↑	see Plankton Community	↑	**WOE Rank 0** -Increased exposure is apparent from WQ and fish tissue chemistry and the elevated strontium concentrations in water and tissue appear to be linked to the Mine. -Fish growth and reproduction responses are mixed - although there are decreases in certain growth and reproduction indices, the pattern of response is not consistent. -Key uncertainties include a possible gear bias in Snap Lake, and uncertainty in aging methods. -Lack of response in fish condition factor, LSI, and GSI and the increase in juvenile growth indicate the degree of impairment, if any, is not ecologically significant.
Fish Tissue Chemistry (Exposure)	Snap Lake Compared to Reference Lakes	↑↑	Thallium exceeds the reference normal range in Snap Lake while the strontium mean in Snap Lake exceeded the reference mean.		
	Snap Lake Compared to Baseline	n/a (no baseline data for small fish)	n/a (no baseline data for small fish)		
Fish Health	Survival	No response	All size classes present in Snap Lake compared to the reference lakes	↓ (uncertain)	
	Growth (Energy Use)	↓	Length and weight, in males and females, and size at age in 2-yr males was lower in Snap Lake than the reference lakes.		
		(↑)	Length/weight in juveniles and size at age for 3-yr females was higher in Snap Lake than the reference lakes.		
	Reproduction (Energy Use)	↓	Egg diameter in females was lower in Snap Lake than the reference lakes.		
		(↑)	Relative fecundity was higher in Snap Lake than the reference lakes.		
	Condition (Energy Storage)	no response	No differences in condition factor or LSI		

Note: The direction of the arrow, up or down, indicates the direction of change or relationship (i.e., increase/positive versus decrease/negative). For biological community structure endpoints, both arrows are included (↑/↓) to reflect that a community shift normally involves combined increases and decrease in abundance and diversity.

↑/↓ = Rating 1; ↑↑/↓↓ = Rating 2. Brackets () indicate that the observed response is not consistent with the hypothesis.

AEMP = Aquatic Effects Monitoring Program; CCME = Canadian Council of Ministers of the Environment ISQG = Interim Sediment Quality Guidelines; GSI = gonadosomatic index; LSI = liver-somatic index; NE = northeast; NMDS = non-metric multidimensional scaling; WQ = water quality; WOE = weight of evidence; TDS = total dissolved solids.

- ***Field Biological Responses –*** For the plankton community in 2012, the zooplankton community exhibited the responses most consistent with the Toxicological Impairment Hypothesis. There was an indication of lower zooplankton biomass in Snap Lake main basin than present at baseline, and this was combined with a species-level community shift. The lower biomass has been evident since 2008, with the exception of 2011. These mild zooplankton responses could be considered consistent with toxicological impairment (resulting in Rating 1 overall for the zooplankton community) but could also be due to top-down (i.e., predation) or bottom up (i.e., food supply) ecological interactions, inter-annual variation or regional factors (e.g., temperature and light).

  Phytoplankton biomass has exhibited a decreasing trend in Snap Lake main basin since 2009 (resulting in Rating 1 for this endpoint) and this has been combined with a shift from a chrysophyceae-cyanobacteria dominated community to a diatom dominated community (resulting in Rating 2 for Community Structure on its own). While a toxicity cause for phytoplankton cannot be ruled out, the responses appear more likely due to enrichment followed by a compensatory community shift, especially given that phytoplankton biomass in Snap Lake remains above baseline, and above biomass observed in Northeast Lake. Considering the influence of this uncertainty on resulting weighting, the phytoplankton endpoint group was judged to be at Rating 1 overall with respect to this hypothesis.

  Combined, the ratings for phytoplankton and zooplankton resulted in Rating 1, overall for the plankton endpoint group.

Integration of the endpoint groups for plankton community exposure and field biological responses indicates that: (i) water quality was altered in Snap Lake in 2012 including multiple parameters which could potentially cause toxicological impairment in the plankton community; and, (ii) concurrent with this, a decrease in zooplankton biomass combined with a species-level shift in the community was also apparent. Given the factors that would mitigate water column toxicity, lack of observed laboratory toxicity, and that the zooplankton response is relatively mild, the strength of evidence for toxicological impairment of the plankton community for 2012 was judged to be at WOE Rank 1.

13.2.2.2 Benthic Invertebrate Community

The endpoint findings and rationale for the rating of each endpoint group for the Benthic Invertebrate Community are as follows:

- ***Exposure:*** The water quality classification for Lake Productivity described above (Rating 1, overall) also applies to benthic invertebrates, but the sediment quality findings in Snap Lake were judged to be more indicative of benthic exposure. In 2012, the most pronounced responses for sediment chemistry were found for bismuth, selenium, sodium, and strontium which each exhibited increasing temporal trends and had concentrations in the main basin that were beyond the baseline normal range. The toxicological significance of these differences was considered uncertain because there are no CCME Interim Sediment Quality Guidelines (ISQGs) or Probable Effect Level (PELs) for these metals, but the trends and differences for these metals indicated potential toxicant exposure resulting in a maximum classification of Rating 2. Rating 2 was also retained overall for exposure endpoint group.

Multiple metals exceeded the CCME ISQG in the Snap Lake main basin (cadmium, chromium, copper, and zinc) but it appears that these concentrations occur naturally in Snap Lake and Northeast Lake, and trends and differences were lacking.

- ***Field Biological Responses*:** The pattern of response in the Benthic Invertebrate Community was a slight decreasing trend in total density and the densities of *Microtendipes* and Pisidiidae, as well as, lower richness in Snap Lake than in Northeast Lake. This trend and difference are consistent with a mild impairment response resulting in a classification of Rating 1 for each, but could also be the result of the decline in phytoplankton biomass and shift in phytoplankton community since 2009 (i.e., changing food supply), top-down interactions (i.e., predation), or inter-annual variability. In the absence of a clear alternative explanation for the mild responses, Rating 1 was applied overall, to the benthic invertebrate community endpoint group.

Integration of the endpoint groups indicates that sediment and water quality have been altered in Snap Lake main basin including multiple parameters that could potentially cause toxicological impairment in the benthic invertebrate and there is a concurrent mild impairment response in the benthic invertebrate community which, in the absence of other influences unrelated to toxicity or enrichment, could be due to toxicant exposure. Given the *a priori* weighting considerations discussed in Section 13.1.4, the mild benthos responses (Rating 1) were judged to best represent the degree of support for toxicological impairment rather than the sediment quality response (Rating 2), resulting in an overall conclusion of WOE Rank 1 for the benthic invertebrate community.

13.2.2.3 Fish Health

The endpoint findings and rationale for the rating of each endpoint group for Fish Health are as follows:

- ***Exposure:*** The water quality classification for the Plankton Community described above (Rating 1, overall) also applies to the fish community. In addition, fish tissue chemistry was anticipated to provide an equal or better indicator of actual exposure to metals because it integrates water quality variations over time, and factors influencing uptake of metals into tissues. For 2012, both mean thallium and strontium[14] tissue concentrations in small-bodied fish from Snap Lake main basin exceeded the reference mean and for thallium the mean was beyond the normal range in the reference lakes. Strontium is known to be elevated in treated effluent and is showing increasing trends in water quality and sediments of Snap Lake but the source of thallium is considered uncertain because it is not showing trends or differences in Snap Lake water and sediment quality.

[14] Note that strontium is known to accumulate in bones of fish and this may have contributed to elevated strontium concentrations in the whole body analyses which included bone. Nevertheless, tissue strontium was found to be higher in Snap Lake Main Basin than the reference lakes.

Therefore, the classification for fish exposure was based on the strontium tissue findings, as well as, the overall water quality conclusion (Rating 1).

- ***Field Biological Responses*:** It is unlikely that fish health has been affected by the changes in water and sediment quality in Snap Lake. Snap Lake fish health endpoints were within the normal range of the pooled reference lakes during 2012, consistent with past conclusions (De Beers 2012). Some fish health endpoints were statistically different between Snap Lake and the reference lakes in 2012 with the differences in the direction that would be consistent with toxicological impairment: lower mean values in Snap Lake compared to the reference lakes for length and weight in males (ages pooled), females (ages pooled), and size at age in 2-yr males; and, lower egg diameter in females in Snap Lake compared to the reference lakes. Consideration of these responses on their own merited a classification of Rating 1 on an individual basis. Conversely, condition was not affected and other endpoints had responses which did not indicate toxicological impairment (e.g., increased fecundity and increased size for juveniles and 3-yr females), suggesting uncertainty in the fish health response as a whole.

  For aquatic organisms, it has been established that juvenile life-stages are typically more sensitive to toxicants than adults (Hutchinson et al. 1998; Mohammed 2013) and also that fecundity is one of the most sensitive indicators of sublethal toxicity (Suter et al. 1987). It follows that if mild toxicological impairment of fish health was beginning in Snap Lake, then effects would first be expected in juvenile fish and the fecundity of adult females. These responses are not being observed, suggesting that the pattern of response in fish health does not support toxicological impairment. Additional uncertainty stems from the possible gear-bias in Snap Lake which may have caused the apparent difference in body sizes between Snap Lake and the reference lakes (refer to Section 7). Based on these considerations, the responses were deemed inconclusive overall, with respect to toxicological impairment, resulting in a classification of Rating 0 for the fish health endpoint group.

Integration of the endpoint groups indicates effects to water quality and resulting tissue chemistry in small bodied fish in Snap Lake, in particular for strontium, but also for multiple water quality parameters that have exceeded AEMP benchmarks and are showing trends and differences. However, fish growth and reproduction responses are mixed, and the overall pattern was not consistent with toxicity since endpoints that are typically considered most sensitive did not have impairment responses. The differences in fish health endpoints measured in 2012 are not greater than changes predicted in the EAR. The EAR predicted that chemicals of potential concern in water and sediment could have a negative effect on fish health, but that the magnitude of this effect would be negligible. No changes to fish reproduction were predicted. There is no direct evidence any of the differences measured in 2012 have affected the ability of fish to survive or reproduce in Snap Lake.

Although exposure of fish to effluent-related substances is occurring, there does not appear to be a toxicological impairment response occurring in fish health, resulting in a WOE Rank 0 for this hypothesis.

13.2.3 Nutrient Enrichment Analysis

The qualitative integration describing the evidence for nutrient enrichment of the Plankton Community and the Benthic Invertebrate Community is summarized in Table 13-10.

Table 13-10 Qualitative Integration for the Nutrient Enrichment Hypothesis

Endpoint Group	Endpoint	Maximum Response Rating	Description of Response	Group Rating	WOE Rank and Rationale
(a) Plankton Community					
Water Quality (Exposure)	Comparison to Benchmarks	no response	-	↑↑	**WOE Rank 2** -WQ exposure indicates nutrient enrichment beyond baseline normal range in Snap Lake. -Phytoplankton response is consistent with enrichment and appears to be at the level of a moderate shift in community structure (i.e., at functional group level) in response to enrichment. -Zooplankton response is not consistent with enrichment.
	Trends in Snap Lake Compared to Reference Lakes	↑	TDS and nitrogen compounds have upward trends relative to Northeast Lake		
	Comparison to Baseline Normal Range	↑↑	TDS and nitrogen compounds have Snap Lake mean above normal range		
Phytoplankton Community (Field Biological Response)	Chlorophyll *a*	no response	-	↑↑/↓↓	
	Phytoplankton Biomass	↑ and (↓)	Increasing trend from 2004 to 2009, decreasing trend back to near baseline levels from 2009 to present.		
	Community Structure	↑↑/↓↓	Shift from chrysophyceae-cyanobacteria dominated community to diatom dominated community.		
Zooplankton Community (Field Biological Response)	Zooplankton Biomass	(↓)	Decrease in Snap Lake compared to baseline		
	Community Structure	↑/↓	Minor community shift observed in NMDS plots		

Table 13-10 Qualitative Integration for the Nutrient Enrichment Hypothesis

Endpoint Group	Endpoint	Maximum Response Rating	Description of Response	Group Rating	WOE Rank and Rationale
(b) Benthic Invertebrate Community (i.e., potential sediment effects)					
Water Quality (Exposure)	Overall Assessment	↑↑	See Plankton Community (above)	↑	**WOE Rank 0** -Enrichment is apparent from water quality. -No clear indication of sediment enrichment (i.e., increased TOC), but some increases in sediment concentrations of nutrients and major ions that are nutrients in the water column. -Water column food supply (primary productivity) was higher in previous years but is now near baseline levels in Snap Lake main basin. -The benthic community response does not appear consistent with nutrient enrichment. -Although *Valvata* density is higher in Snap Lake, the trend is downward suggesting that this is not an enrichment response but could be a natural difference.
Sediment Quality (Exposure)	Comparison to Reference Lakes and Baseline	↑	Nitrate, TKN, TN are higher in Snap Lake main basin than in Northeast Lake		
	Temporal Trends	↑	Available potassium and sulphate have increasing trends in Snap Lake main basin		
Primary Productivity (Exposure)	Chlorophyll *a* and phytoplankton biomass	↑ and (↓)	Phytoplankton biomass trend - see Plankton Community (above)		
Benthic Invertebrate Community (Field Biological Response)	Total Density	(↓)	Slight decreasing trend in Snap Lake compared to Northeast Lake	no response	
	Richness	(↓)	Lower richness in Snap Lake than Northeast Lake		
	Diversity	no response	-		
	Evenness	↑	Decreasing evenness trend in Northeast Lake but evenness remains relatively constant in Snap Lake		
	Density of Dominant Taxa	(↓)	Decreasing trend in Snap Lake compared to Northeast Lake, for *Microtendipes* and Pisidiidae		
		↑	*Valvata* density is higher in Snap Lake relative to Northeast Lake, but trend in Valvata density is slightly downward		
	Community Structure	no difference	-		

Table 13-10 Qualitative Integration for the Nutrient Enrichment Hypothesis

Endpoint Group	Endpoint	Maximum Response Rating	Description of Response	Group Rating	WOE Rank and Rationale
(c) Fish Community (i.e., potential water column effects)					
Water Quality (Exposure)	Overall Assessment	↑↑	See Plankton Community (above)	↑↑	**WOE Rank 0 (uncertain)** -Enrichment is apparent from water quality and phytoplankton, but not evident in food supply (zooplankton and invertebrates). -Fish growth and reproduction responses are mixed, but on balance could also possibly indicate enrichment. -No response in fish condition factor, LSI and GSI indicates the degree of enrichment, if any, is mild.
Primary Productivity (Exposure)	Chlorophyll *a* and phytoplankton biomass	↑ and (↓)	Phytoplankton biomass trend - see Plankton Community (above)	↑ and (↓)	
Food Supply (Exposure)	Zooplankton biomass	(↓)	Decrease in Snap Lake compared to baseline	no response*	
	Benthic invertebrate total density	(↓)	Decreasing trend in Snap Lake compared to Northeast Lake	no response*	
Fish Health	Survival	no response	All size classes present in Snap Lake compared to the reference lakes	↑ (uncertain)	
	Growth (Energy Use)	↑	Length/weight in juveniles and size at age for 3-yr females was higher in Snap Lake than the reference lakes		
		(↓)	Length and weight in males and females, and size at age in 2-yr males was lower in Snap Lake than the reference lakes		
	Reproduction (Energy Use)	↑	Relative fecundity was higher in Snap Lake than the reference lakes		
		(↓)	Egg diameter in females was lower in Snap Lake than the reference lakes		
	Condition (Energy Storage)	no response	No differences in K or LSI		

Notes: The direction of the arrow, up or down, indicates the direction of change or relationship (i.e., increase/positive versus decrease/negative). For biological community structure endpoints, both arrows are included (↑/↓) to reflect that a community shift normally involves combined increases and decrease in abundance and diversity.
Brackets () indicate that the observed response is not consistent with the hypothesis.
*no response consistent with hypothesis
↑/↓ = Rating 1; ↑↑/↓↓ = Rating 2; TN = total nitrogen; TDS = total dissolved solids; TKN = total Kjeldahl nitrogen; NMDS = non-metric multidimensional scaling; WOE = weight of evidence; LSI = liver-somatic index; GSI = gonado-somatic index; TOC = total organic carbon; K = condition factor; WQ = water quality.

13.2.3.1 Plankton Community

The endpoint findings and rationale for the rating of each endpoint group for the Plankton Community are as follows:

- ***Exposure:*** Water quality in Snap Lake is the main indicator of exposure for the plankton community. Snap Lake is expected to be phosphorus-limited and total phosphorus (TP) does not show an increase Snap Lake. However, phosphorus dynamics in Snap Lake are poorly understood (Section 5), so the lack of response for TP may not be indicative of a lack of nutrient enrichment, for 2012. Nutrient enrichment in Snap Lake was indicated by increasing trends and concentrations beyond the normal range for TDS (including calcium which can be a nutrient for zooplankton and benthic invertebrates) and nitrogen compounds (nitrite, nitrate, and ammonia). In addition, total Kjeldahl nitrogen (TKN) had an increasing trend but is within the Snap Lake normal range. The concentrations of TDS and nitrogen compounds beyond the normal range classify as Rating 2 for these parameters and this rating was also applied overall for the exposure endpoint group.
- ***Field Biological Responses:*** Enrichment of the phytoplankton community appears to be happening as indicated by the trends in phytoplankton biomass and community shift in the phytoplankton community. Phytoplankton biomass exhibited an increasing trend from 2004 to 2009, followed by a decreasing trend back to near baseline levels from 2009 to the present. This biomass trajectory has been combined with a shift from a chrysophyceae-cyanobacteria dominated community to a diatom dominated community (resulting in Rating 2 for Community Structure). The likely explanation for these changes is an enrichment-caused biomass increase followed by a compensatory community shift that then reduced biomass. In contrast, the pattern of response in zooplankton (decreased biomass relative to baseline) did not appear consistent with enrichment, but this does not outweigh the conclusion that enrichment is happening in the phytoplankton community. A possible explanation for the lack of an apparent enrichment response in the zooplankton community is predation pressure.

  Based on these considerations, Rating 2 was applied overall for the plankton endpoint group to represent the functional group shift and biomass increase in phytoplankton.

Integration of the endpoint groups indicates that there is evidence of nutrient increases in the water column combined with a pattern of response in the phytoplankton community at a moderate level, based on the shift from chrysophyceae-cyanobacteria to diatoms. These findings are consistent with an overall WOE Rank of 2 (moderate support) for the Nutrient Enrichment Hypothesis for the Plankton Community.

13.2.3.2 Benthic Invertebrate Community

The endpoint findings and rationale for the rating of each endpoint group for the Benthic Invertebrate Community are as follows:

- ***Exposure:*** The water quality classification for the Plankton Community described above that indicates enrichment with nitrogen compounds and TDS (Rating 2, overall) also applies to benthic invertebrate exposure, but measures of potential sediment enrichment and increased food supply were also considered to represent benthic exposure to nutrients. Nitrogen compounds were elevated in Snap Lake main basin sediments compared to Northeast Lake - these parameters are not direct nutrients for benthic invertebrates but indicate a potential enrichment "signature" in the water column. In contrast, total organic carbon (TOC) is naturally high in Snap Lake and Northeast Lake with no apparent differences between the two. With regard to food supply for benthic invertebrates, phytoplankton biomass (food supply for filter feeders) reached a peak well above baseline in 2009, but has declined back to near baseline suggesting previous but not current enrichment of food supply (no difference relative to baseline). Thus, although water quality and sediment quality indicate chemical enrichment of Snap Lake, the 2012 monitoring did not indicate higher food supply relative to baseline conditions. Based on these findings Rating 1, overall, was considered an appropriate representation of the exposure endpoint group.
- ***Field Biological Responses*****:** The pattern of response in the benthic invertebrate community was generally that of mild decreasing trends or endpoints that are slightly lower in Snap Lake than in Northeast Lake, and this pattern was not consistent with that expected under a response to nutrient enrichment. Although *Valvata* density was higher in Snap Lake for 2012 (resulting in Rating 1 for this individual endpoint), the trend has been slightly downward since 2009 suggesting that higher density in Snap Lake is not an enrichment response but could be a natural difference. Thus, for 2012, the benthic invertebrate community was rated as no response, overall, with respect to the Nutrient Enrichment Hypothesis.

Integration of the endpoint groups indicates chemical enrichment of water and sediments in Snap Lake, which has influenced the phytoplankton community but did not result in a higher biomass of phytoplankton food supply for benthic invertebrates in 2012. Also, there is little indication that the detrital food supply in sediments (i.e., total organic carbon) has increased, although TOC is naturally high in Snap Lake. The pattern of response in the benthic invertebrate community is not consistent with that which would be expected in response to nutrient enrichment. Given the *a priori* weighting considerations discussed in Section 13.1.4, the lack of consistent benthos response combined with lack of increased food supply for benthic invertebrates is considered indicative of little support for the Nutrient Enrichment Hypothesis, resulting in an overall conclusion of WOE Rank 0 (hypothesis not supported).

13.2.3.3 Fish Health

The endpoint findings and rationale for the rating of each endpoint group for Fish Health are as follows:

- ***Exposure:*** The water column and phytoplankton considerations discussed for exposure endpoints for benthic invertebrates (i.e., chemical enrichment of the water column but no current increase in the food supply for benthos and zooplankton) also apply to the fish community. In addition, neither of the densities of zooplankton and benthic invertebrates that are the direct food supply for fish were indicative of increased food supply for fish (i.e., the response in these endpoints was of decreasing trends, and/or lower biomass in Snap Lake than Northeast Lake). Thus, Rating 1 overall was considered to represent current exposure conditions which include chemical enrichment of the water column, but no increased food supply for fish.

- ***Field Biological Responses:*** As discussed for toxicological impairment, Snap Lake fish health endpoints were within the normal range of the pooled reference lakes during 2012. However, some fish health endpoints were statistically different between Snap Lake and the reference lakes in 2012 with the differences in the direction that would be consistent with nutrient enrichment: larger body size for juveniles and 3-yr females; and, increased relative fecundity in females. Conversely, condition and liver-somatic index (LSI) were not affected, and other endpoints had responses that were not consistent with nutrient enrichment: lower mean values in Snap Lake compared to the reference lakes for length and weight in males (ages pooled), females (ages pooled), and size at age in 2-yr males; and, lower egg diameter in females. Individually, these mixed endpoint responses resulted in contradictory classifications of Rating 1, either in support of, or not in support of the nutrient enrichment hypothesis, but no consistent response. Therefore, the responses were deemed inconclusive, overall, with respect to nutrient enrichment, resulting in a Rating of 0 for the fish health endpoint.

Integration of the endpoint groups indicates chemical enrichment of the water column in Snap Lake, which has influenced phytoplankton community structure but did not result in a higher food supply of zooplankton and benthic invertebrates for fish in 2012. The fish health endpoint responses were mixed and, on balance, did not indicate an enrichment response in 2012. Given that measurement of field biological responses is expected to be the strongest indicator of actual environmental effects, the lack of a clear fish health response, combined with no indication of increased food supply for fish were considered indicative that support for the nutrient enrichment hypotheses is lacking, resulting in an overall conclusion of WOE Rank 0 (hypothesis not supported).

13.3 QUALITATIVE INTEGRATION SUMMARY

Both hypotheses regarding the nature of possible effects in Snap Lake were potentially supported based on the results of the 2012 AEMP.

For the Toxicological Impairment Hypothesis, the results of the qualitative integration of exposure and field biological responses resulted in the following WOE Rankings:

- ***Plankton Community*** – WOE Rank 1
- ***Benthic Invertebrate Community*** – WOE Rank 1
- ***Fish Health*** – WOE Rank 0

differences from the normal range in water quality (chloride, fluoride, nitrate, and TDS), sediment quality (bismuth, selenium, sodium, and strontium), and fish tissue chemistry (primarily strontium). Biological responses consistent with toxicological impairment were a mild decrease in zooplankton biomass combined with a species-level community shift, and mild decreases or decreasing trends in benthic invertebrate biomass and richness. The nature of these responses is mild and within the range of variability that might also be expected from ecological interactions such as changing predation pressure or changes in food supply, or inter-annual variability. Therefore, the classification of Rank 1 for plankton and benthic invertebrates is considered conservative. The responses of the phytoplankton community and fish health were generally not consistent with this hypothesis.

Thus, the conditions in Snap Lake for 2012 provided a weak indication that toxicological impairment responses may be occurring in zooplankton and benthic invertebrates, but in all cases the responses were considered mild. Future AEMP monitoring will provide information necessary to determine whether this is an actual progressive impairment in response to Mine activities, or whether it is due to causes unrelated to the Mine.

For the Nutrient Enrichment Hypothesis, the results of the qualitative integration of exposure and field biological responses resulted in the following WOE Rankings:

- ***Plankton Community*** – WOE Rank 2
- ***Benthic Invertebrate Community*** – WOE Rank 0
- ***Fish Health*** – WOE Rank 0

Increased exposure to potential nutrients by these biological components of the Snap Lake ecosystem was indicated by increasing trends or differences from the normal range in water quality (TDS and nitrogen compounds) and sediment quality (nitrogen compounds). For phytoplankton, the biomass trajectory (increases until 2009 and then decreases) combined with the moderate level community shift, appears to be consistent with nutrient enrichment, resulting in the moderate level of support for the Nutrient Enrichment Hypothesis by the plankton community. However, for the remaining biological components of Snap Lake (zooplankton, benthic invertebrates, and fish), there was very little evidence of enrichment-related responses in terms of increased food supply or biological response endpoints.

The AEMP findings for Snap Lake for 2012 provided moderate evidence of for enrichment of the phytoplankton community, but provided little support that this enrichment was extending to higher trophic levels. The lack of response in the higher trophic levels could possibly be explained by the shift in the phytoplankton community (i.e., biomass is near baseline levels) or by a concurrent mild impairment response, as described above, which might counter-act any enrichment response. On-going AEMP monitoring and special studies are expected to provide an improved understanding of nutrient and productivity dynamics in Snap Lake.

response. On-going AEMP monitoring and special studies are expected to provide an improved understanding of nutrient and productivity dynamics in Snap Lake.

It is important to note that this analysis represents a “snap-shot” of potential responses in Snap Lake resulting from treated effluent release from the Mine and that variations in the degree and nature of responses can be anticipated from year to year. For 2012, the most prominent Mine-related effect in Snap Lake appeared to be changes to water and sediment quality, combined with enrichment of the phytoplankton community and a resulting community shift. The remaining biological responses in the zooplankton and benthic invertebrate communities were mild in all cases, and there was no consistent response in fish health. Based on these findings it can be concluded that although there was weak to moderate support for each hypothesis for certain ecosystem components, there has been no impairment of the structure and function of the Snap Lake ecosystem through 2012.

13.4 REFERENCES

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14 ACTION LEVELS

The Action Levels/Response Framework of the Snap Lake Aquatic Effects Monitoring Program (AEMP) will be reviewed by regulators and the Mackenzie Valley Land and Water Board (MVLWB) in May and June 2013. Upon approval by the MVLWB, the Response Framework of the AEMP will be initiated. Any Action Levels that are exceeded will be reported to the MVLWB and in 2013 and subsequent Annual Reports.

15 RECOMMENDATIONS

Where available, each section of the 2012 Aquatic Effects Monitoring Program (AEMP) provided recommendations for consideration. These recommendations are detailed below.

Section 2 - Site Characterization and Supporting Environmental Variables

- The AEMP report should continue to review and consider spills and incidents which have the potential to affect the aquatic environment. Year-to-year changes to the project 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 to capture variations in spring temperatures. Redundancy should be built into the temperature logger program to verify data and reduce potential loss of data from field error and equipment malfunction.
- Ice thickness measurements should be extended to Lake 13 and continued for Snap Lake and Northeast Lake.
- De Beers site staff should continue to take descriptive and accurate notes related to ice cover on Snap Lake.
- Hydrological measurements should continue to be collected to record the peak of freshet.

Section 3 - Water Quality

Data Quality and Continual Improvement

- Implement the recommendations from the quality assurance (QA) and quality control (QC) assessment (outlined in Appendix 3A), which focus on investigating potential contamination, variability between samples, and alleviating holding time issues. These include 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 investigate the accuracy and precision of analyzing individual components of total phosphorus by the analytical laboratories currently used in the AEMP program (Section 12.4). Continued split sampling, which is part of the regular QA/QC procedures in the AEMP, is recommended to provide an external check of the primary laboratories completing the analyses. A limited number of nutrient spike samples should routinely be sent to several laboratories as an on-going and independent check of the accuracy of nutrient results.

Water Quality Data Interpretation

- Review the application of the Canadian Council of Ministers of the Environment fluoride, chloride and nitrate Water Quality Guidelines because there are known ameliorating factors that would apply in Snap Lake. Proposed site-specific benchmarks and management actions for nitrate and total dissolved solids (TDS) (which includes chloride and fluoride) are currently under development for the Snap Lake Mine as part of the Nitrogen and TDS Response Plans, respectively. In accordance with the Water Licence (MVLWB 2012), these plans will include a description of the sources of nitrogen and TDS, a description of the ecological implications of nitrogen and TDS loadings on the receiving environment, and a discussion on options for reducing loadings.

- Give consideration to the four parameters with concentrations that have increased beyond the normal range in Snap Lake, but for which there are no relevant AEMP benchmarks (i.e., barium, lithium, rubidium, and strontium). A separate Response Plan for strontium is being prepared; however, it is recommended that for the remaining parameters (total barium, lithium, and rubidium), available toxicological literature be reviewed to assess the implications of increases in these parameters.

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 related to increased loadings.
- Sulphate was not identified as a key parameter during the most recent lake model update because the Canadian Council of Ministers of the Environment do not currently provide Water Quality Guidelines for sulphate. Sulphate will be included in future lake model updates, because flow-weighted concentrations in the effluent were above environmental assessment report predictions.
- Concentrations of antimony should continue to be investigated through the follow-up QA/QC measures listed above. If the QA/QC investigation indicates that the observed values above the maximum acceptable concentrations are real, total antimony should be added to the modelling parameter suite to investigate whether any physical or chemical processes are influencing antimony concentrations and what the maximum concentrations throughout the lake are predicted to be throughout Mine operations.
- Re-visit the acidification assessment completed in 2009 using updated air modelling information and water quality data to determine whether the results from the 2009 assessment remain valid.

Study Design

- After completion of the acidification re-assessment, the water quality sampling program in the inland lakes should be re-visited to determine whether the current design is appropriate.
- Shift the focus from spatial and seasonal trends in Snap Lake to changes downstream of Snap Lake. As the overall water quality begins to change in Snap Lake, the seasonal and spatial differences in water quality in the lake become less relevant and the temporal changes in Snap Lake and changes downstream of Snap Lake become more relevant. In response to the changes in water quality, the number of monitoring stations in Snap Lake should be reduced; information gathered from the Downstream Lakes Special Study should

be used to establish new downstream AEMP stations in addition to the current KING01 station.

- Because the results of nutrient samples collected at different depths demonstrate that nutrient concentrations, particularly total phosphorus, may vary with sampling depth, reduction to a single combined sampling depth for water quality and plankton components is not recommended at this time (Section 12.4). Additional nutrient samples should be collected at mid-depth and in the euphotic zone to better define which forms of nutrients differ with sample depth and the degree to which this difference may affect other nutrient-related components and activities at Snap Lake, such as benthic invertebrates and water quality modelling. The results should be reported jointly as part of an “eutrophication indicators section” in the AEMP report.

Section 4 - Sediment Quality

- Continue to use Northeast Lake as a reference lake to assess long-term regional trends.
- Repeat sediment quality sampling in Lake 13 in 2013 to determine whether the elevated and variable concentrations of arsenic, barium, and manganese observed in 2012 are representative of actual conditions in Lake 13. The suitability of Lake 13 as a second reference lake for sediment needs to be confirmed prior to the next full round of AEMP sediment quality monitoring, which under the proposed AEMP Design Update would take place in 2015.

Section 5 - Plankton

- Increased silica concentrations may be allowing for greater growth of diatoms. Data on silica should be collected as part of the plankton component of the AEMP rather than solely as part of the water quality component of the AEMP. This would provide depth-integrated samples that could be directly compared to samples collected during the plankton program, and would result in a better understanding of the quantity of silica that is available to diatoms in both lakes.
- An underwater light meter should be used in addition to Secchi depth to measure light penetration into the water column. Light penetration may be a major variable affecting plankton that needs to be measured with less uncertainty than with a Secchi disk.
- Additional evaluation should be undertaken of the suitability, in terms of plankton metrics, of Lake 13 as an appropriate second reference lake.

Section 6 – Benthic Invertebrate Community

- The benthic invertebrate community monitoring program should be conducted again in 2013 to determine whether the decreasing trends in total density, Microtendipes density, Pisidiidae density, and richness in 2012 continue. This recommendation is made because the direction of the effect observed in previous years has reversed in 2012.

Section 7 – Fish Health

- The size of fish from Downstream Lake 1 is consistent with the size of fish from Snap Lake. A future fish health program inclusive of Downstream Lake 1 to monitor fish health and to assess the extent of Mine influence on fish populations in the downstream lakes is proposed. It is recommended that a fish population survey be performed in Downstream Lake 1 prior to the next fish health program to determine whether similar fish populations and species are present and provide baseline information on lake ecology.
- Given the lack of reliable aging data from laboratory analyses, it is recommended that laboratory aging be excluded from future fish health programs until and unless an age validation study for Lake Chub is completed. It is recommended that the full non-lethal sample size (i.e., up to 400 fish, inclusive of fish size measurements) continue to be targeted to allow for age-determination based on length-frequency distributions.
- In subsequent re-evaluations of the fish health component, otolith age data from past reports should be converted to the assigned age used in the 2012 AEMP such that comparisons of size at age and between adults and juveniles are consistent.
- Future studies should review the normal range of fish endpoints within the Snap Lake main basin over time and between pooled references lakes. An assessment of the sensitivity of the normal range comparisons versus the Critical Effects Size comparisons should be undertaken for the Snap Lake Mine towards determining the significance of an effect to fish health.
- Stomach content analyses allow for consideration of potential differences in diet among lakes when interpreting Lake Chub health endpoints such as condition and relative liver size. Further, stomach content analyses allow integration of conclusions between the benthic invertebrate survey and potential consequences to fish health in Snap Lake. It is only possible to predict what changes in the benthic invertebrate community might have on fish species if the importance of individual invertebrate species, genera, or families as a prey item for fish is understood. It is recommended fish stomach content analyses be continued in future fish health programs.
- Liver triglyceride and glycogen endpoints offer valuable supporting information when interpreting differences in fish condition, relative liver size, and reproduction. The liver triglyceride and glycogen endpoints were added due to increasing concentrations of nutrients and TDS and possible indicators of nutrient enrichment in Snap Lake, and should continue to be measured in future fish health programs. Further, the number of liver samples analyzed from each lake should be increased to improve statistical power.
- Because size bias as a result of gear selectivity may be occurring in the study lakes, it is recommended that similar levels of effort by each fishing method be expended in each lake in future programs, regardless of initial fishing successes. While it is likely that differences in the success of various gear types will remain among lakes, a more balanced fishing effort with each gear type is required to determine whether a gear bias exists for size or sex of fish collected among the lakes.

- Further to expending comparable amounts of time on the various fishing methods among lakes, it is recommended that the proportion of fish captured that are in spawning condition by each fishing method be calculated to examine whether reproductive status is comparable among fishing methods and among areas. Fish that will spawn will be defined as fish that are found to be pre-spawning, ripe, or spent. It is also recommended that the sex ratio of adult fish from Snap Lake and the reference lakes be examined in future programs to determine whether a different number of males and females are caught by different gear types in different lakes. If the sex ratio is not influenced by fishing method, sex ratios should be examined to determine whether differences exist in sex ratios among lakes or whether potential activities of males and females differ at the time of capture (e.g., spawning versus feeding).

Section 9 – Fish Tissue Chemistry

- This study represents the first small-bodied fish tissue survey in Snap Lake. This survey should continue every three years to provide an early indicator of potential changes in large-bodied fish and to provide data that may assist in interpreting cause(s) of any potential effects observed during the small-bodied fish health survey.
- As the source of increased thallium concentration in Lake Chub collected from Snap Lake is uncertain, trends over time in thallium concentrations for Lake Trout and Round Whitefish should be reviewed during the 2013 large-bodied fish tissue chemistry survey.

Section 12.1 – Littoral Zone Special Study

- Additional training of divers is recommended prior to sampling: in-water training and evaluation; at least one of the divers scientifically trained; and, resampling if initial sampling is not conducted properly.
- Duplicate samples should be collected to allow for an estimate of analytical variance.
- A rapid microscope assessment should be conducted of the lake water samples from both Snap Lake and Northeast Lake to assess the presence of plankton.
- One more sampling station should be added in the northwest arm of Snap Lake to increase spatial coverage through the gradient of effluent exposure present in this part of the lake. And two more sampling stations should be added in Northeast Lake to increase statistical power and spatial coverage.
- A more detailed taxonomic evaluation is recommended, i.e., providing the taxonomist used in 2004 a subset of samples from 2013 to identify and enumerate.
- Light in the water column should be measured using underwater light meters along with examination of the attenuation coefficient at each littoral sampling station.
- A 250 µm mesh sieve should be used for collection of littoral zone biota.
- Artificial substrates should be deployed for invertebrate sampling, in combination with sweep net sampling. Using artificial substrates would allow quantitative sampling of littoral zone

invertebrates, including attached taxa, while sweep net sampling would allow documenting the full diversity of the littoral zone invertebrate community.

Section 12.2 – Downstream Lakes Special Study

- Monitoring in the downstream lakes (DSL1 and DSL2) and Lac Capot Blanc should continue to evaluate the dispersion of treated effluent associated with Snap Lake Mine discharge.
- Downstream water quality predictions should be revisited as additional data are collected.
- A hydrodynamic downstream model should be developed using the three dimensional hydrodynamic and water quality model that was used to model water quality in Snap Lake. This model would be used to predict TDS concentrations at various points in the downstream lakes including near the inlet and outlet, in deep pockets, and as whole-lake averages. The initial focus of the model would be the first three downstream lakes (i.e., DSL1, DSL2, and Lac Capot Blanc), but could be expanded in future as necessary and appropriate.
- The temperature logger program should install temperature loggers earlier in the year to capture spring temperature variations. Redundancy should be built into the temperature logger program to verify data and reduce the potential loss of data from field error and equipment failure. It is recommended that one of the shallow site temperature loggers be installed as close to the inlet stream as possible for modelling.
- Additional bathymetric surveys should be conducted in the east basin and southern portions of Lac Capot Blanc to fill in data gaps and to provide more detailed bathymetry maps of the lake.

Section 12.3 –Reference Lake 13 Suitability Special Study

- Lake 13 should be included in future AEMP sampling as a second reference lake for Snap Lake.
- Additional monitoring of the effects of the winter road (e.g., water, snowpack, dust) on Lake 13 should be included in the 2014 AEMP.

Section 12.4 – Nutrient Special Study

- The results of the laboratory assessment indicated that there is no clear choice for a preferred analytical laboratory because the percent error results from the three laboratories (University of Alberta, Maxxam Analytics, and ALS Canada Ltd. [ALS]) in the spike samples did not differentiate any one laboratory in terms of overall accuracy. Therefore, ALS should continue as the primary laboratory for the water quality component of the AEMP. The results of the split samples reinforce this recommendation, because ALS provided the most consistent nutrient data, with the fewest notable differences to results from the other two laboratories.
- Continued split sampling, which is part of the regular QA/QC procedures in the AEMP, is recommended to provide an external check of the primary laboratories completing the analyses. A limited number of nutrient spike samples should routinely be sent to the three

laboratories as an on-going and independent check of the accuracy of nutrient results from all three laboratories.

- The lack of accuracy in low-level phosphorus results in the spike samples should be considered when interpreting trends in phosphorus data in Snap Lake, establishing management action levels for phosphorus, and developing nutrient models for Snap Lake. Existing efforts to reduce uncertainty in low-level phosphorus, such as reducing field contamination of samples through documented QA/QC procedures, should continue.
- Because the results of nutrient samples collected at different depths demonstrate that nutrient concentrations, particularly total phosphorus, may vary with sampling depth, reduction to a single combined sampling depth for water quality and plankton components is not recommended at this time. Additional nutrient samples should be collected at mid-depth and in the euphotic zone to better define which forms of nutrients differ with sample depth and the degree to which this difference may affect other nutrient-related components and activities at Snap Lake, such as benthic invertebrates and water quality modelling.

References

MVLWB. 2012. Mackenzie Valley Land and Water Board Licence #MV2011L2-0004. Yellowknife, NWT, Canada.

16 CLOSURE

This report was prepared by the undersigned, and reviewed by Alexandra Hood, Environmental Superintendent, De Beers Snap Lake Mine. The Littoral Zone Special Study was also reviewed by Dr. Michael Turner of Fisheries and Oceans Canada; Golder and De Beers are grateful for the contributions and suggestions of Dr. Turner. Golder and De Beers would like to thank site staff including Gail Seto and Erin Rowlands (Environmental Coordinators), Tom Bradbury (Permitting Coordinator), Melissa Leclair, Andrea Hrynkiw, Guylaine Gueguen and Andre Boulanger for field logistic support, running the fish tasting activities and site information. A special thanks as well to Terry Kruger and Sabet Biscaye for facilitating the fish tasting activity and subsequent reporting. Addition00sal Golder staff contributed to this report; we gratefully acknowledge the following contributors: Robin Bourke and Chris Madland (Site Characterization and Supporting Environmental Variables); Giovanna Diaz, Cherie Frick, Jonathon Love, Louise Simpson, and Alison Snow (Water Quality); Shevelle Hutt (Plankton); Clayton James and Melanie Jaeger (Fish Health); Leslie Carroll (Fish Tissue Chemistry); Alison Humphries (Nutrient Special Study); Joel Farah and Carmen Walker (Drafting and GIS); Elodie Taniere, Ada Ma, Mark Jaferllari, and Tatiana Leclerc (Report Production); and, Karin Lintner (Formatting, Editing, and Report Production).

We trust the above meets your present requirements. If you have any questions or require additional details, please contact the undersigned.

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